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Evaluating the effectiveness of seabird bycatch mitigation measures for pelagic longlines in the South Atlantic

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ABSTRACT

Incidental mortality (bycatch) of seabirds in pelagic longline fisheries remains a major threat to many populations. The design and implementation of technical innovations aimed at reducing seabird bycatch rates have long been a focus of research. However, it has historically been difficult to extrapolate the efficacy of a particular mitigation measure to the scale of seabird populations or oceanic basins. Here, we develop an ecological risk assessment for five populations of threatened albatross and petrel species that forage in the south Atlantic Ocean. Since seabird bycatch rates are likely under-reported to fisheries regulatory bodies, we adopted a risk-based approach to predict differences in bycatch rates between different combinations and specifications of mitigation measures, comparing those currently specified by the International Commission for the Conservation of Atlantic Tunas (ICCAT) against best practice guidelines recommended by the Seabird Bycatch Working Group of the Agreement on the Conservation of Albatrosses and Petrels (ACAP). We conclude that updating existing mitigation measure specifications for pelagic longlining in the South Atlantic to reflect current best practice guidelines would potentially reduce seabird mortality by 41–86 %, compared to use of any two of the three options by vessels. Simultaneous application of all three mitigation measures recommended as current ACAP best practice was predicted to reduce seabird mortality by 72–93 % and therefore should be considered by ICCAT as the most appropriate management measure for seabirds until further data are available to undertake more rigorous analyses.

1. Introduction

Fishing activity poses the greatest at-sea threat to seabirds globally (Dias et al., 2019; Gee et al., 2023), with pelagic longlining accounting for a considerable proportion of the total incidental mortality (bycatch) (Huang, 2011; Tuck et al., 2011; Jiménez et al., 2020; Votier et al., 2023). In pelagic longlining, most seabird bycatch occurs during setting, when birds become hooked or entangled in branch lines, and drown as these lines sink, although in some fisheries, substantial numbers are also caught live during hauling, many of which die from their injuries (da Rocha et al., 2021; Phillips and Wood, 2020; Jiménez et al., 2014). Several seabird bycatch mitigation measures are proven to be effective, including branch line weighting to increase hook sink rate (e.g. Melvin et al., 2013; Jiménez et al., 2019a; Gilman et al., 2025), setting gear at night when seabirds are less active or less likely to see bait (e.g. Brothers et al., 1999; Jiménez et al., 2020; Kroodsma et al., 2023), and 'bird-scaring' (streamer or tori) lines that deter birds from entering the risk area astern of the vessel (e.g. Melvin et al., 2013; Rollinson et al., 2016; Jiménez et al., 2020). There have also been more recent technological developments of mitigation measures such as hook shielding devices, which enclose the hook barb until it reaches a pre-determined depth (Sullivan and Barrington, 2021).

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The Seabird Bycatch Working Group of the Agreement on the Conservation of Albatrosses and Petrels (ACAP) periodically reviews the evidence base for these measures and publishes advice on best practice for the selection and specification of appropriate measures (ACAP, 2023; Pierre, 2023). However, implementation of best practice varies widely (Baker et al., 2024).

Official reporting of seabird mortality at sea are not representative of all fleets, and data are generally inadequate for conservation (e.g., even if a bird is correctly identified to species level, the population from which it originated will be unknown). There have been several field trials of different mitigation measures (e.g., Melvin et al., 2013; Rollinson et al., 2016; Jiménez et al., 2019b; Sullivan and Barrington, 2021) but individual studies are not considered representative of fisheries that span ocean basins. It is therefore necessary to consider alternate methodologies that integrate the best available scientific information at a scale relevant to regional fisheries management (Reid et al., 2023). Regional Fisheries Management Organisations (RFMOs), such as the International Commission for the Conservation of Atlantic Tunas (ICCAT), have a responsibility to ensure that their fleets reduce bycatch of seabirds, as well as other threatened taxa like turtles or sharks. Seabird bycatch mitigation measures that apply to pelagic longlining in the south Atlantic Ocean are detailed in International Commission for the Conservation of Atlantic Tunas (ICCAT) Recommendations (Rec) 07–07 and 11–09:

- Per Rec 11–09, in areas south of 25°S, at least two of the following measures shall be applied:
  1. No setting of lines between nautical dawn and nautical dusk. Deck lighting to be kept to a minimum;
  2. Bird-scaring (tori) line(s) are deployed during setting (specification varies, dependent upon vessel length); or
  3. Branch line weights are deployed (minimum weight varies, depending on distance between weight and hook).
- Per Rec 07–07, in areas between 20 and 25°S, bird-scaring lines at least must be deployed (with certain exceptions).

Current ACAP best practice advice (ACAP, 2023) is that pelagic longline fishing vessels either:

- Implement night setting, with branch line weighting, and bird-scaring lines, to the specification in ACAP (2023) during all sets, or
- Implement hook shielding devices, configured to open after the device has reached 10 m depth or been in the water >10 min, for all hooks.

We evaluated performance of different combinations of mitigation measures using the 'ecological risk assessment of the sustainability of fisheries' (EASI-Fish; Griffiths et al., 2019) to integrate best available information for the south Atlantic from ICCAT, ACAP, and academic sources. Using a proxy for fishing mortality, we evaluated performance of different combinations of mitigation measures for five populations of four seabird species (Atlantic Yellow-nosed Albatross, Tristan Albatross, Wandering Albatross, and White-chinned Petrel; Fig. 1) that breed on one or more of the UK Overseas Territories (UKOTs) in the southern Atlantic. The UKOTs of South Georgia and the South Sandwich Islands, Tristan da Cunha, and the Falkland Islands are key nesting sites for several globally threatened seabird species (Fig. 1).

2. Materials and methods

2.1. Fisheries data

To calculate the overlap between seabird foraging distribution and fishing activity in the ICCAT Convention Area (south of 20°S), this study uses spatial data to estimate the extent and distribution of effort by ICCAT pelagic longline fleets from ICCAT (EffDis) (Taylor et al., 2020), and Global Fishing Watch (GFW) (Kroodsma et al., 2018), for the period between 2012 and 2020. EffDis is the total estimated effort (in # of hooks per quarter) set per 5° x 5° cell for each fleet.

The model included seven individual fleets and one group of fleets representing mean fishing effort in the Atlantic south of 20°S between 2012 and 2020. The distribution of 'major' fleets (Chinese Taipei, Japan, Spain, Brazil, South Korea, Namibia, and South Africa; comprising 75.3 % of the total fishing effort), plus a 'minor' fleet grouping of all other flag state vessels was to estimate overlap with seabird foraging distribution.

Although pelagic longlines typically extend many tens of kilometres, the risk area for seabird bycatch is close to the vessel itself during setting of lines whilst the hooks are at or near the surface. Consequently, the spatial resolution of GFW data (1° x 1°), bounded by the extent of EffDis (5° x 5°), is more suited to understanding these finer scale interactions. The fishing footprint used here also includes vessels (predominantly flagged to Japan and Korea between 40 and 45°S) targeting southern bluefin tuna *Thunnus maccoyii* (Fig. 2), in the ICCAT Convention Area, but that is otherwise managed by the Convention for the Conservation of Southern Bluefin Tuna (CCSBT). These were included in our risk assessment since they are required to comply with ICCAT measures whilst fishing in the Atlantic.

Some model information used in previous applications of EASI-Fish (IATTC, 2023), for instance maximum setting depth of longline hooks, was unnecessary in the present study. White-chinned petrels *Procellaria aequinoctialis* are the deepest diving of the species reviewed here, reaching a maximum reported depth of 22 m, and capable of diving at up to 2.0 m.s^{-1} (Frankish et al., 2021a; Rollinson et al., 2014). Albatrosses typically only dive at most to 5–8 m (Bentley et al., 2021), although exceptionally to as much as 19 m (Guilford et al., 2022). The shallowest reported hook setting depth of pelagic longlines among the papers reviewed for vessels active in the south Atlantic was 50 m (18 m float line plus 32 m branch lines; Afonso et al., 2012), meaning that birds are only vulnerable to hooking whilst lines are being set or hauled close to the vessel.

2.2. Seabird data

The spatial distribution of each seabird population, summed across breeding and nonbreeding seasons, was taken from Carneiro et al. (2020). Data were available for all species and populations at 5° x 5° resolution either quarterly or as an annual average. The annual average distribution (covering the period 2007–2016) was used here for five populations of four species (Table 1; Fig. 3). These species are all listed by ACAP and as globally threatened by the International Union for the Conservation of Nature (IUCN). They were selected for our study as they show substantial overlap with pelagic longline activity in the south Atlantic and are commonly recorded as bycatch in those fisheries (Bugoni et al., 2008; Jiménez et al., 2011, 2020). The four species reviewed here comprised 47 % of the total bycatch (from among 28 species overall) reported in the south Atlantic by Jiménez et al. (2020). White-chinned petrels are especially vulnerable to bycatch in the south Atlantic and elsewhere (Jiménez et al., 2020; Phillips et al., 2006; da Rocha et al., 2021).

Overlap was calculated by fleet and species as the proportion of overlap between polygons of fishing effort and bird at-sea distributions, ignoring the lowermost 5 % of observations to reduce influence of rarely used areas (data from ICCAT, and Carneiro et al., 2020). Seasonal patterns in fishing effort and bird distribution were not considered since relative effort was already accounted for in other model parameters. Bird distribution at sea (Carneiro et al., 2020), and fishing effort (apparent hours fished, 1° x 1°, following Kroodsma et al., 2018) were clipped to a polygon including 95 % of the distribution of fishing effort (EffDis), and then we calculated the proportion of each that was within the remaining area.

Biological data for each species were collated from a literature search. Unless known, standard deviation in parameter estimates was set

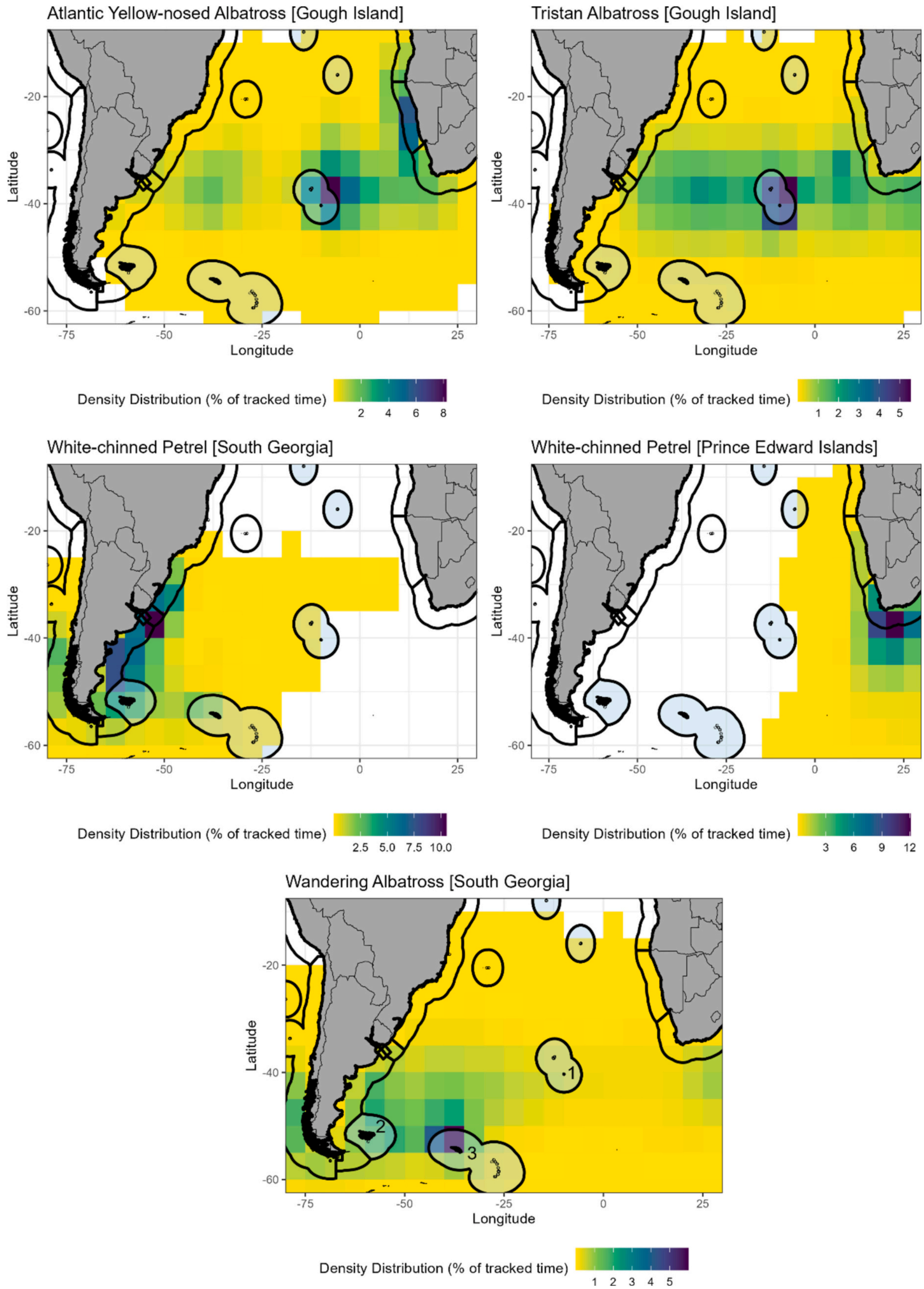

Fig. 1. Distribution of the five seabird populations included in our study (data from Carneiro et al., 2020) and their breeding site(s) in United Kingdom Overseas Territories (UKOT). UKOTs (shaded areas) that host globally important populations of seabirds are highlighted on the bottom panel: 1 = Gough Island, part of the Tristan da Cunha islands and the territory of Ascension, St Helena, and Tristan da Cunha; 2 = Falkland Islands; and 3 = South Georgia, part of the territory of South Georgia and the South Sandwich Islands. EEZs (thick black lines) are those available from marineregions.org and do not necessarily represent boundaries agreed in the course of bi- or multilateral negotiations.

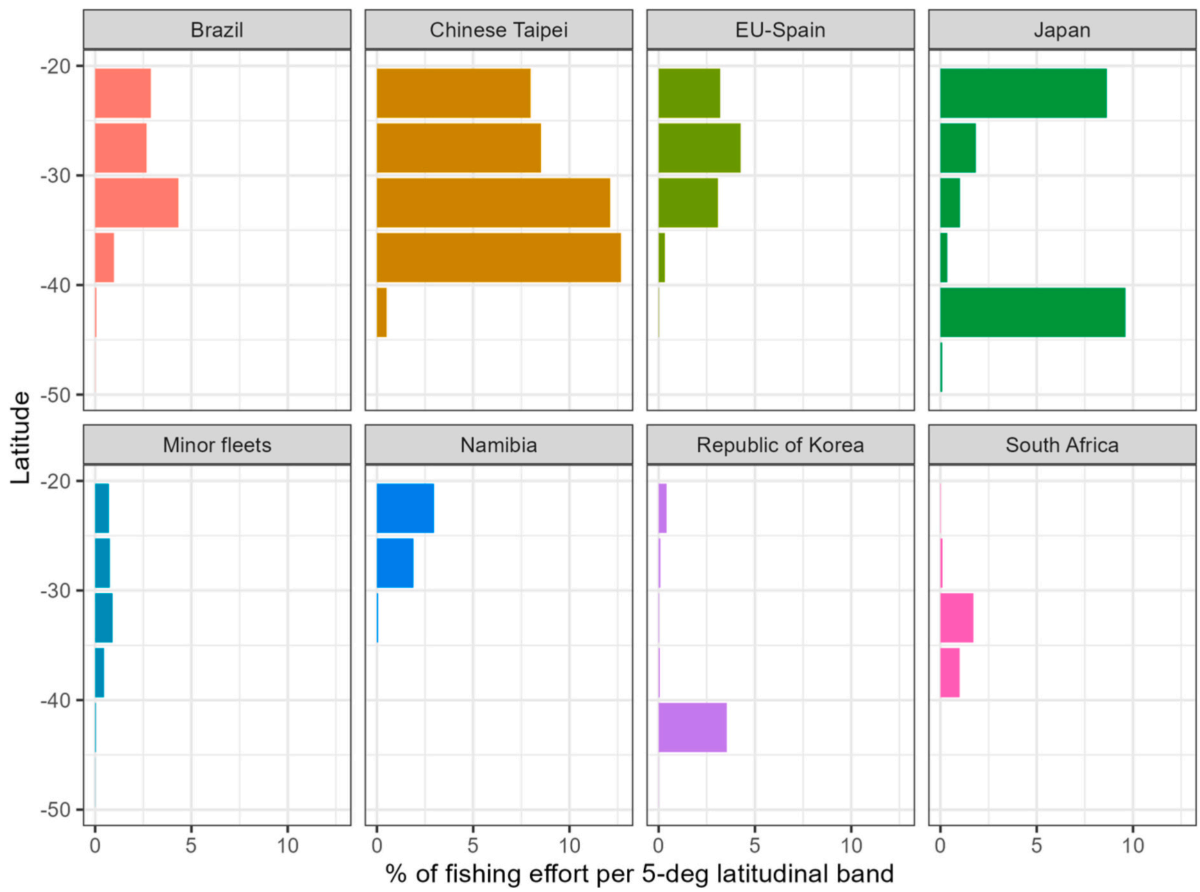

Fig. 2. Latitudinal distribution of fishing effort (% of total per fleet and 5° latitudinal band) south of 20°S in the ICCAT convention area per flag state or flag state group (ICCAT EffDis, 2012–2020).

at 10 % of the mean value (distributions shown in Appendix 1). Seabirds achieve their asymptotic body size by the time of fledging (ca. 3–9 months after hatching, depending on species). Birds of all age classes except chicks at the colony are therefore vulnerable to capture in fisheries, with the relative susceptibility dependent on ontogenetic differences in foraging range and behaviour (e.g., Frankish et al., 2020; Frankish et al., 2021b; Gianuca et al., 2017; Gimeno et al., 2022). These are challenging to constrain directly given the limited tracking data from juvenile and immature birds (Carneiro et al., 2020).

2.3. Mitigation measures

We reviewed the mitigation measures currently mandated by ICCAT (line weighting, night setting, and bird-scaring lines), comparing them to equivalent measures as specified by ACAP (2023). We also assessed hook shielding devices (HSDs) as an alternative measure (Table 2). Currently, fleets are required to use two of three measures when fishing south of 25°S (i.e. following any of Status Quo scenarios SQ1–3 in Table 2), or bird-scaring lines when fishing between 20 and 25°S (ICCAT Rec 07–07). We evaluated a set of nine scenarios applied to all fishing south of 20°S or 25°S (Table 2).

ICCAT Rec 11–09 and ACAP best practice (ACAP, 2023) for night setting do not differ substantively (Appendix 2) and so estimates of bycatch rates related to night setting were applied equally across the different scenarios.

Given the lack of operational information, this model assumes full compliance with the measures as specified within each scenario. Presuming the parameter estimation otherwise accurately represents species vulnerability to bycatch, assuming perfect compliance means the model likely underestimates at-sea bycatch rates. Any estimated difference in seabird mortality thus arises primarily from the degree to which the different specifications of these measures (ICCAT and ACAP) influence real-world bycatch rates. Kroodsma et al. (2023) showed that compliance with night setting measures is generally low (3–5.5 %) but current ICCAT recommendations require two out of three measures (ICCAT Recs 07–07 and 11–09) and so do not mandate night setting. Data on the implementation of bird-scaring lines or branch line weighting are unavailable for the ICCAT Convention Area, meaning that the evidence of Kroodsma et al. (2023) is insufficient to adjust the implementation of this model.

Finally, we provide an evaluation of the relative performance should Recs 07–07 and 11–09 be combined into a single set of measures applied to the entire ICCAT Convention Area south of 20°S. However, we note that the majority of the observed distribution of the five populations in this study (Table 1; Fig. 3) occurs south of 25°S (Carneiro et al., 2020).

2.4. Bycatch model specification

We use an age-structured implementation of the EASI-Fish model (Griffiths et al., 2019; IATTC, 2023). This model was selected for its suitability to data-limited settings; in this case where species-specific observations of seabird interactions are likely to be under- or misreported and, in the case of rarer species, likely to be insufficient for robust analysis (Brothers et al., 2010; Morkūnas et al., 2022). The EASI-Fish approach was selected because it does not require empirical data on bycatch rates and instead estimates a proxy for instantaneous fishing mortality rate (F yr^{-1}) from the product of a series of parameters relating to the overlap between fishing activity and at-sea distribution, and the proportion of the seabird population that is susceptible to capture, including error distributions (Table 3). The estimate of F is effectively

Table 1
Summary information on the species and populations modelled in this study. *Endemic to that island group. Atlantic-yellow-nosed albatrosses *Thalassarche chlororhynchos* breed elsewhere at Tristan da Cunha but data from Gough are considered representative of all breeding populations.

Species	Breeding location	Population size estimate of breeding birds	**IUCN Status** (of species) *Trend* (of population size; BirdLife International, 2024)
Atlantic yellow-nosed albatross (*Thalassarche chlororhynchos*)	Gough Island, Tristan da Cunha (40° S, 10° W)	35,000–73,000 (Ryan et al., 2011, Bratt, 2023)	**Endangered** Declined at Gough Island between 2008 and 2014 (3–5.6 % yr^{-1}), followed by a partial recovery from 2014 to 2020 (0.9–4.1 % yr^{-1}) and likely decreasing elsewhere (1.1–5.0 % yr^{-1})
Tristan albatross (*Diomedea dabbenena*)	Gough Island, Tristan da Cunha* (40° S, 10° W)	3000–4000 (Oppel et al., 2022)	**Critically Endangered** Declining (1.0–1.2 % yr^{-1})
Wandering albatross (*Diomedea exulans*)	South Georgia (54° S, 37° W)	2600 (Poncet et al., 2017)	**Vulnerable** Declining (1.4–4.1 % yr^{-1})
White-chinned petrel (*Procellaria aequinoctialis*)	South Georgia (54° S, 37° W)	1 180,000–2,370,000 (Martin et al., 2009)	**Vulnerable** Declining (1.6–1.9 % yr^{-1}) at South Georgia. No estimate of population trend for the Prince Edward Islands.
	Prince Edward Islands (47° S, 38° W)	9000–15,000 (Ryan et al., 2012)	

the residual likelihood of mortality within a given period, once all other relevant and estimable parameters have been discounted. EASI-Fish includes functionality for estimating biological reference points such as the ratio of F to the value of F at maximum sustainable yield (F_{msy}) to determine the vulnerability status of the population (Griffiths et al., 2019). However, determination of vulnerability status was not required in this study as the goal was to understand which specification of mitigation measures demonstrated the greatest potential for minimising F.

F-at-age rates per population were calculated per fleet as follows, tailoring the EASI-Fish approach (Griffiths et al., 2019) to the specific terms relating to seabird susceptibility to pelagic longlining (Eqs. 1–3; with parameters given in Table 3):

Eq. 1 (finite fishing mortality):

$$\text{finiteF} = \sum\nolimits_{\text{fleet},\left(\frac{max.age}{\text{nclass age}}\right)} s^*o^*a_{seas}{}^*a_{spat}{}^*e^*avm^*((1-avm)^*prm) \quad (1)$$

Eq. 2 (adjusted finite fishing mortality):

$$\text{adj.finiteF} = \sum\nolimits_{\text{fleet}}(\text{finiteF}^*q)^*E \quad (2)$$

Eq. 3 (instantaneous fishing mortality):

$$Inst.F = \sum\nolimits_{fleet} -log(1-adj.finiteF) \quad (3)$$

Estimates of F were subsequently reported as the mean value across the total age range of each population (Eq. 1), to account for differences in selectivity and maturity at age. The model was structured in 0.5 year age classes (hence, finite f per fleet of a species with a longevity of 20 years would be the mean of 40 estimates of finite F-at-age).

Seabirds are wide-ranging and can travel 100 s of km per day. The spatial resolution of fleet activity (1° x 1°, 85.5 km^2 at 40°S) was greater than the attraction distance for albatrosses attending fishing vessels (up to 30 km, Collet et al., 2015; Kroodsma et al., 2023). Catchability (*q*, gear efficiency) was estimated using the 'domain of potential interaction' (Griffiths et al., 2007) that uses gear length and animal movement characteristics to approximate the effective fishing area where birds overlap with vessels. Here, the rate at which birds attended a vessel within a given cell was calculated as the length of the set (typically 90-100 km, Brothers et al., 1999; Bugoni et al., 2008; Afonso et al., 2012; Melvin et al., 2013, 2014; Fernandez-Carvalho et al., 2015) multiplied by the attraction radius around a vessel (30 km), resulting in a mean *q* of 0.82 (i.e. the area of attraction of a single set, and therefore chance a bird will attend, is on average 82 % of the area over which fishing effort is gridded). This was further adjusted by a multiplier representing the proportion of time that vessels typically spend setting gears (typically 6.5 h +/− 1.5 s.d.; Tuck et al., 2003; Melvin et al., 2013).

The main mechanism by which the model outputs differ between scenarios is through the estimation of the encounterability parameter (e), which is expressed as a function of the combination of mitigation measures applied in each scenario. To estimate encounterability per scenario (Table 2) we collated the available information on seabird interaction rates from the scientific literature. To reduce the influence of publication bias, we compiled all papers listed in the reviews conducted by Pierre (2023) and ACAP (2023), as well as papers listed in a Web of Science search using the following search string: *(seabird* OR "sea bird" OR "sea birds") AND (bycatch OR "by-catch" OR "incidental catch*" OR "incidental capture*") AND (mitigat* OR prevent* OR reduc*) AND long-lin**. Finally, we cross-referenced all relevant tuna-RFMO meeting documents. Following the compilation of papers, we retained those in which:

I) The study evaluated the performance of bycatch mitigation measures through comparison of at least two treatments, so that relative performance (including comparisons to no mitigation) could be assessed,
II) Specifications of bycatch mitigation measures were adequately detailed and followed either current ACAP best practice (ACAP, 2023) and/or ICCAT, 2011 standards,
III) Reported interaction rates allowed for standardization (e.g., reported bycatch/ contact/ attack rates per unit effort), and
IV) Sample sizes of each treatment were reported in a standardised fashion (i.e., 1000 s of hooks) so that studies could be weighted.

Ultimately, 27 suitable studies were used to estimate interaction rates (Table 3). Of these, 14 were conducted in the South Atlantic, representing 41.9 million hooks (70 % of the total of 60.0 million hooks across all studies).

2.5. Estimating interaction rates

Empirical observations of interaction rates are difficult to compare between studies because of a range of factors, including the relative abundance behind vessels and the composition of the species assemblage (Jiménez et al., 2011; De la Cruz et al., 2022), and the area and season in which fishing occurred. It was therefore necessary to reduce these observations to second-order values that reflect relative differences within individual studies.

Here we define a term for encounterability as the 'relative Standardised Interaction Rates' (rSIR) (Table 3). We estimated rSIR for each mitigation measure combination within a Bayesian framework to account for the influence of different sample sizes among studies and to propagate uncertainty around the estimates. Specifically, we first standardised the reported interaction rates per unit effort (IPUE) for each bycatch mitigation treatment. IPUE was used instead of the more

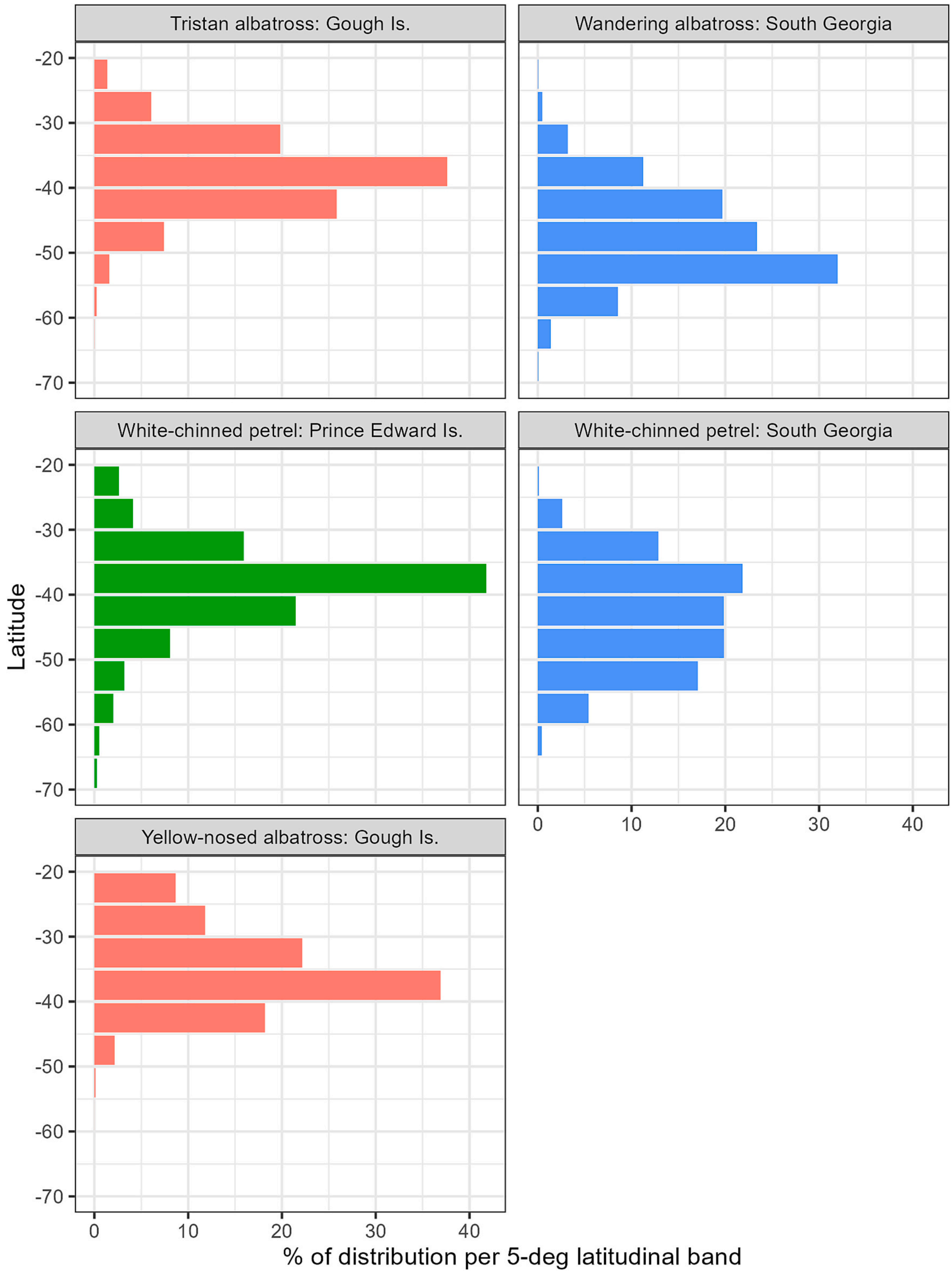

Fig. 3. Average annual distribution (% of total) by latitude per seabird population within the Atlantic. All populations range elsewhere in the Pacific or Indian Oceans (Fig. 1).

common measure, bycatch per unit effort, to allow inclusion of a broader range of studies in the estimation of rSIR:

Estimating relative standardised interactions

$$rSIR_{i,j} = \frac{IPUE_{ij}}{max_i(IPUE_j)}, \tag{4}$$

in which, $IPUE_{i,j}$ refers to the interaction rate per unit effort (bycatch/contact/attack rate, usually per 1000 hooks) per mitigation measure (i) and trial (j). If a paper reported adequately on several studies or trials that differed considerably (e.g., studies in different ocean basins; Sullivan et al., 2017), these were considered separately in our analyses. Through this approach, the treatment (mitigation measure) with the highest reported IPUE per study received an rSIR of 1, and each other treatment in the same study was scaled accordingly. Due to this

Table 2
Summary of mitigation measure scenarios (bycatch mitigation requirements) considered in this study. All scenarios are for fishing south of 20° and 25°S. Full specifications of in Appendix 2. SQ = Status Quo (mitigation measures implemented to current ICCAT Recommendations); ACAP = mitigation measures implemented to ACAP Best Practice specifications).

Scenario	Mitigation measures applied	Specification of measures per:
SQ1	Branch line weighting Night setting	ICCAT Rec 07–07 ICCAT Rec 11–09
SQ2	Branch line weighting Bird-scaring lines	
SQ3	Night setting Bird-scaring lines	
SQ4	Night setting Branch line weighting Bird-scaring lines	Extension of ICCAT Recs 07–07 and 11–09 but maintaining ICCAT specifications
ACAP1	Branch line weighting Night setting	Updating ICCAT Recs 07–07 and 11–09 to ACAP best practice advice (2023) specifications
ACAP2	Branch line weighting Bird-scaring lines	
ACAP3	Night setting Bird-scaring lines	
ACAP4	Night setting Branch line weighting Bird-scaring lines	ACAP best practice advice (ACAP, 2023)
ACAP5	Hook shielding devices	

standardization approach, the relationship between rSIR and IPUE was not linear across scenarios—the more mitigation measures applied, the less scope for improvements to bycatch rates. Despite this challenge, our approach leveraged, rather than was limited by the context-specificity of individual bycatch mitigation studies, and allowed for subsequent evaluation of relative performance of different measures and specifications across studies.

We fitted Bayesian GLMMs with a stochastic node to the collated data to estimate rSIR per mitigation measure under ACAP best practice (ACAP, 2023) or ICCAT (Rec 07–07 and 11–09) specifications (Eqs. 5-6).

Eqs. 5 & 6 – GLM specification fitting to rSIR observations

$$\left(rSIR_{i,j}{}^{*}n_{hooks_{i,j}}\right) \sim Binomial\left(rSIR_{i,s}, n_{hooks_{i,j}}\right), \tag{5}$$

$$Logit\left(rSIR_{i,s}\right) = \alpha_i + \theta^{\beta}_{i,s}{}^{*}spec_{i,j} + \varepsilon_{ocean,i,j} + \varepsilon_{data,i,j}, \tag{6}$$

in which $rSIR_{i,s}$ is the relative Standardised Interaction Rate per bycatch mitigation measure (i) with specifications (s), α_i is the intercept, $\theta^{\beta}_{i,s}$ is a vector of β coefficients for the fixed effects of mitigation measure (i) with specifications (s), $spec_{i,j}$ is the design matrix of the relevant specifications of mitigation measure (i) in study (j), $\varepsilon_{ocean,i,j}$ is a random effect accounting for ocean-basin level differences (South Atlantic, South Indian, South Pacific, or North Pacific Ocean), $\varepsilon_{data,i,j}$ is a random effect accounting for the two fundamentally different data types included (bycatch or contact/attack rates), and $n_{hooks\ i,j}$ is the sample size per mitigation measure (i) in study (j) in 1000 s of hooks. To ensure that no individual study could dominate estimates, we restricted the maximum value to 500,000 hooks. Our modelling approach was not informative (i.e., credible intervals 0–1) for bycatch mitigation measures evaluated by a single study, and thus we excluded the only suitable study of underwater bait setters (Robertson et al., 2018). This approach allowed us to account specifically for different sample sizes and thus the potential different levels of confidence in the available evidence, while also generating uncertainty as appropriate around the rSIR estimates.

Table 3
Calculation of parameters used to estimate incidental mortality (bycatch) of seabirds in ICCAT fisheries. GFW = Global Fishing Watch. For implementation, see Eqs. 1–3.

Parameter	Definition/ calculation	Source of information
Effort (*E*)	Relative longline effort per flag state (hooks set, 2012–20) as a proportion of the most active fleet	EffDis
Overlap (*o*)	Overlap of polygons (convex hull from raster layers gridded at 5° x 5°) containing 95 % of ICCAT fishing activity and bird distribution.	EffDis; Carneiro et al., 2020
Spatial availability (a_{spat})	Relative apparent fishing effort and seabird foraging time within overlapping area, using 1° x 1° GFW data and 5° x 5° tagging data respectively	Kroodsma et al., 2018; Carneiro et al., 2020
Catchability (*q*)	Proportion of fishing effort cell (1° x 1°, GFW) covered by 'attraction area' which was taken as length of longline x max. Attraction distance (30 km).	Afonso et al., 2012; Brothers et al., 1999; Bugoni et al., 2008; Collet et al., 2015; Fernandez-Carvalho et al., 2015; Gales et al., 1998; Griffiths et al., 2007; Melvin et al., 2013, 2014
Encounterability (*e*)	Proportion of birds interacting with hooks, as a function of the mitigation measures applied per scenario, bird diving depth, and typical setting time derived from relative standardised interaction rate (rSIR) information estimated from published IPUE values.	Baker and Wise, 2005; Baker and Candy, 2014; Boggs, 2001; Brothers et al., 1999; Domingo et al., 2017; Duckworth, 1995; Gales et al., 1998; Gianuca et al., 2011, 2021; Gilman et al., 2008, 2023; Goad et al., 2019; Jiménez et al., 2012, 2019a, 2019b, 2020; Klaer and Polacheck, 1998; Melvin et al., 2013, 2014; Meyer and MacKenzie, 2022; Petersen et al., 2008; Robertson et al., 2018; Rollinson et al., 2016; Santos et al., 2019; Sato et al., 2013, 2014; Sullivan et al., 2017; Gilman et al., 2007; Jiménez et al., 2012; Lokkeborg., 2003; Katsumata et al., 2015; Minami and Kiyota, 2002; Minami et al., 2011; Ochi, 2022, 2023; Ochi et al., 2013; Chaloupka et al., 2021; Gales et al., 1998; Gilman et al., 2025; Baker et al., 2016
At-vessel mortality (*avm*)	Proportion of hooked seabirds (*Diomedea*, *Thalassarche*, or *Procellaria* spp.) killed in ICCAT pelagic longline fisheries. Mean value = 0.96 (+/− s.d. 0.0103)	ICCAT, 2023
Post-release mortality (*prm*)	Proportion of hooked seabirds released alive that die subsequently. Mean value = 0.40 (+/− s.d. 0.10)	Phillips and Wood, 2020

To extend our rSIR estimates for individual mitigation measures to the combinations of mitigation measures (with either ACAP best practice or ICCAT Rec 07–07 and 11–09 specifications), we calculated the product of the relevant $rSIR_{i,s}$ estimates. This step was required as there was not a sufficiently large enough sample size of studies/trials evaluating each combination of mitigation measures with varying specifications (e.g., Pierre, 2023). Following the estimation of rSIR for individual mitigation measures and combinations thereof, we also calculated the relative gains that could be achieved when changing existing ICCAT (Res 07–07 and 11–09) specifications, for the Atlantic south of 20°S, to ACAP best practice specifications.

We fitted our rSIR model within the Bayesian modelling programme OpenBUGS (Spiegelhalter et al., 2014). Specifically, we used vague priors only ($\alpha_i \sim N[0, 0.001]$, $\beta \sim N[0,1]$, $\sigma_\varepsilon^{-2} \sim U[0,10]$) and fitted models using three MCMC chains of 150,000 iterations following a burn-in of 75,000 iterations. We assessed convergence by evaluating trace plots visually and by confirming that $\widehat{R}$ <1.05. We report our estimates as medians with 95 % credible intervals (CIs) unless otherwise stated.

2.6. Parameter estimation and sensitivity testing

For each iteration, parameters (Table 3) were resampled from within a beta distribution (Sinharay, 2010) with a fixed standard deviation (set at 10 % of their mean where error unknown). The final set of model solutions was a product of at least 25,000 iterations or continued until the standard error of instantaneous F had converged (most recent 1000 iterations +/− 0.2 % of all previous iterations). Larger solution sets (up to 50,000 iterations) were trialled, but did not improve precision.

Given that the only material difference between scenarios, and upon which the conclusions of this study rely, is the estimation of encounterability, sensitivity testing was restricted to this parameter only. Other parameters vary by fleet or population, but mean values and error rates were fixed between scenarios. We examined the impact of the 50 % over- and under-estimation of all encounterability parameters, using model solution sets of 5000 iterations.

Seasonal availability was not explicitly estimated to avoid duplicating error estimation with other parameters (E and a_{spat}; Table 3). Firstly, overall effort (E) weights fleets according to their activity estimated by ICCAT, meaning that fleets with lower mean annual effort were of lower importance for bycatch estimation. This is inclusive of fleets which may be highly seasonal such as fishing for southern bluefin tuna by Japanese and Korean vessels between 40 and 45°S (Fig. 2). Secondly, the available tagging data permitted spatial availability to be estimated such that it is weighted against areas where distributions overlapped but that were of minor importance for either birds or vessels.

3. Results

3.1. Performance of individual measures

Branch line weighting according to ACAP best practice guidelines had a median rSIR of 0.139 (95 % CI range: 0.017–0.440; Fig. 4), compared with an interaction rate under ICCAT specifications of 0.507 (95 % CI range: 0.099–0.833; Fig. 4). Performance of bird-scaring lines was similar between ACAP and ICCAT (rSIR CI ranges: 0.065–0.759 and 0.052–0.711 respectively; Fig. 4). Night setting requirements were the same for both ACAP and ICCAT (rSIR 95 % CI ranges: 0.010–0.325; Fig. 4). Hook shielding devices as a stand-alone measure had a rSIR of 0.069 (95 % CI range: 0–0.296; Fig. 4).

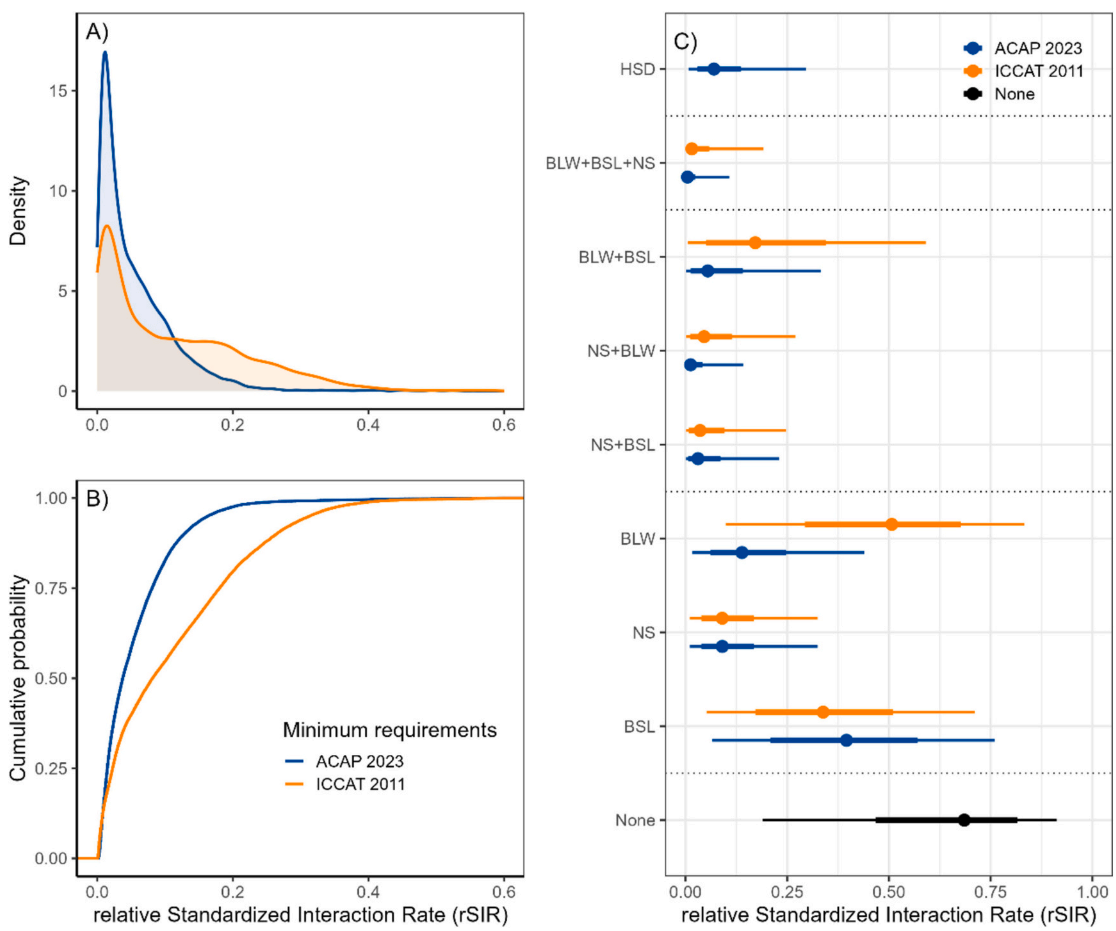

Fig. 4. Density plots (A) and cumulative density functions (B), and relative performance (rSIR) of individual and combined mitigation measures (C) under ICCAT recommendations, and ACAP best practice. The density plot (A) illustrates the distribution of the MCMC iterations of the rSIR estimates, similar to a histogram. If cumulative density functions (B) do not cross, first order stochastic dominance exists, and the specifications with the lower rSIR estimate is indeed the better performing, despite uncertainty. BLW = Branch Line Weighting; BSL = Bird-scaring Lines; HSD = Hook Shielding Devices; NS = Night Setting.

3.2. Best performing combinations of current mitigation measures

Similarly performing scenarios were grouped across species approximately as follows in Fig. 5. The best performing combinations of mitigation measures under ICCAT or ACAP specifications were all three measures (SQ4 and ACAP4) or branch line weighting and night setting to ACAP specifications (ACAP1).

Given the uncertainty in the estimation of the various interaction terms, there was considerable overlap in the estimated bycatch mortality between several scenarios. Scenarios ACAP2–3, SQ1, SQ3, and ACAP5 were generally similar across species (Average Performance in Table 4). Scenarios that included night setting had smaller differentials between ICCAT and ACAP specifications, since these requirements do not differ.

Atlantic yellow-nosed and Tristan albatrosses *Diomedea dabbenena* from Gough Island, and white-chinned petrels from South Georgia were the most vulnerable to bycatch (Fig. 5). Without attempting to validate against observer reports, bycatch rates (for breeding birds) under current measures were estimated at 1000–4000 Atlantic yellow-nosed albatrosses, and 22,600–90,900 white-chinned petrels from South Georgia per year. For both populations however, mortality estimates under the best performing scenarios (ACAP4 and ACAP1) were much lower: between 300 and 500 Atlantic yellow-nosed albatrosses and < 10,000 white-chinned petrels from South Georgia per year. Across all species, implementing all three measures to ACAP specifications represented a

Table 4
Seabird bycatch mitigation measure combinations grouped into specifications (columns) and tiers of performance (rows) in terms of reducing seabird bycatch mortality on pelagic longlines. Numbers in [] denote scenario.

Scenario	ICCAT (2011)	ACAP (2023)
Best performance	SQ4 – All existing measures applied simultaneously	ACAP1 – Night setting and branch line weighting ACAP4 – All existing measures applied simultaneously
Average performance	SQ1 – Night setting and branch line weighting SQ3 – Bird-scaring lines and night setting	ACAP2 – Bird-scaring lines and branch line weighting ACAP3 – Bird-scaring lines and night setting ACAP5 – Hook shielding devices
Worst performance	SQ2 – Bird-scaring lines and branch line weighting	

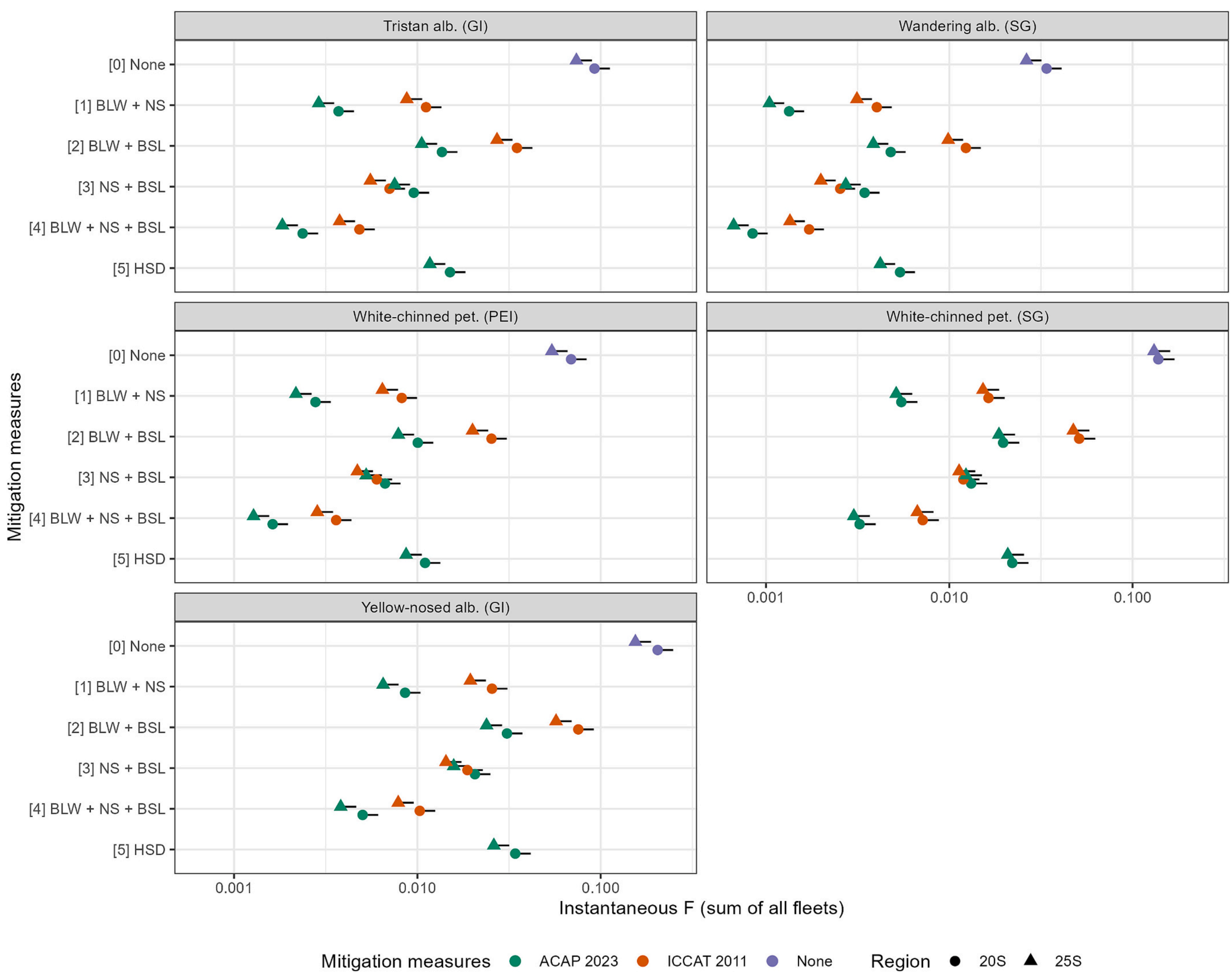

Fig. 5. Estimated instantaneous fishing mortality rates per population, and scenario (mean ± 95 % confidence intervals). Region = Model spatial domain either south of 20°S or south of 25°S. Mitigation measures = specification of mitigation measures as per ICCAT (Rec 07–07 and 11–09) or ACAP (2023). GI = Gough Island; PEI = Prince Edward Islands; SG = South Georgia. Numbers in [] refer to scenario numbers (see Table 2). BLW = Branch Line Weighting; BSL = Bird-scaring Lines; HSD = Hook Shielding Devices; NS = Night Setting.

reduction in bycatch mortality of 89–94 % versus the worst performing, currently-mandated combination of mitigation measures (bird-scaring lines and branch line weighting; SQ2), or around a 98 % reduction versus fishing with no mitigation measures in place.

3.3. Relative gains

Updating the mitigation measure specifications from currently mandated (SQ1–3) to ACAP (2023) guidelines (ACAP1–3) resulted in a mean reduction in bycatch rates across all species of 16–67 % (Table 5). Adopting all three measures to ACAP specifications (ACAP4) was estimated to reduce bycatch rates by 72–93 % against perfect compliance with existing measures (Table 5).

When including fishing activity between 20 and 25°S, significantly higher mortality in all scenarios was estimated for all populations except white-chinned petrels from South Georgia (Fig. 5).

3.4. Sensitivity

In the EASI-Fish framework, estimation of mortality rates is via the product of independent parameters; hence sensitivity was proportional to its magnitude, and relative sensitivity was consistent across species and scenarios (Appendix 2). Sensitivity to encounterability values was consistent across scenarios, and proportional to the magnitude of the estimate of F. Varying encounterability parameters by a factor of 0.5 and 1.5 resulted in relative differences in total estimated F of 0.733–0.759, and 1.482–1.533 respectively (Appendix 3). Sensitivity to parameters is thus only expected to impact our estimates substantially if there are systematic differences in the reporting of published observer data for mitigation measure as specified by ICCAT or ACAP, which could not be addressed via the estimation of rSIR.

4. Discussion

Our models compare predicted bycatch totals for five seabird populations in pelagic longline fisheries in the south Atlantic Ocean between the mitigation measures currently mandated by ICCAT and current best practice specifications recommended by ACAP (2023). Our analysis used the EASI-Fish approach (Griffiths et al., 2019), combined with novel modelling approaches to estimate interaction rates. Actual seabird bycatch rates or totals cannot be estimated directly because the performance of different combinations of bycatch mitigation will differ across the fishery. Nevertheless, ours is the first study that aims to determine how alternative combinations of mitigation measures and specifications would influence seabird bycatch rates at a scale relevant to tuna RFMOs.

4.1. Comparisons with at-sea observations

Globally, seabird bycatch from longlining is estimated as 160,000 to 320,000 birds per year (Anderson et al., 2011), including 41,000 albatrosses and petrels caught in the southern hemisphere by pelagic longliners (Abraham et al., 2019). There is concern that under-reporting of seabird bycatch is widespread (e.g. Brothers et al., 2010; Morkūnas et al., 2022). Bycatch rates reported to fisheries commissions often do not faithfully represent species composition, and cannot be scaled to estimate total bycatch, and several States whose vessels conduct pelagic longlining in the south Atlantic reported no bycatch during any observed trips (ICCAT ST09 data, 2019–2021). Evident disparities between direct observer reports and official ICCAT data underline this uncertainty (Jiménez et al., 2020; ICCAT, 2023). Given the mean observed bycatch rate from published studies in the South Atlantic (0.86 birds/1000 hooks; Gianuca et al., 2011, 2021; Jiménez et al., 2014, 2020, Melvin et al., 2014; Sullivan et al., 2017; Robertson et al., 2018), and the number of hooks set between 2019 and 2021 (112.5 million hooks; ICCAT EffDis), the expected annual bycatch is approximately 32,300 seabirds. Bycatch totals cannot be estimated for most fleets in the ICCAT region because of a lack of data. Between 2019 and 2021 (and notwithstanding the extent to which normal patterns of fishing activity were influenced by the COVID pandemic) 1457 interactions of seabirds with fishing gear were reported by observers on Japanese, Brazilian, Spanish and South African pelagic longline vessels, covering between 2 and 10 % of trips (mean 6 %). Of these interactions, 91 % were reported as the bird “discarded dead” (ICCAT ST09 data). Annual mortality from these fleets (51 % of total longline effort in the same period), would therefore be approximately 8100 seabirds between 2019 and 2021, or 15,900 assuming bycatch rates on unobserved vessels were similar to those on the vessels with observers. However, there are disparities in terms of species composition; white-chinned petrels accounted for around 6 % of total bycatch recorded by ICCAT flag states, compared with 45–84 % of captures in Jiménez et al. (2020) and da Rocha et al. (2021). The disparities between observed versus reported data are the primary reason for adopting the EASI-Fish approach where mortality can be estimated as a product of relevant interaction probabilities. This includes the rate for species that are rarely recorded, for which bycatch rates are challenging to estimate from on-board observer data.

Jiménez et al. (2020) reported 8472 captures in total of 28 species on board vessels flagged to Brazil, Portugal, South Africa, Japan and Uruguay. In total, 37.2 million hooks were observed between 2002 and 2016, from 583 fishing trips. Between 2012 and 2020, an average of 38.7 million hooks were set annually by all fleets in the south Atlantic (EffDis). Of the captures reported in Jiménez et al. (2020), 3842 (45 %) were of white-chinned petrels which we assume were solely from South Georgia (as birds from the Prince Edward Islands use waters in the southeast Atlantic, and their population is approximately 1 % of the size; Fig. 1; Table 1). Using our estimated mortality rates for the same fleets, bycatch rates according to current combinations of mitigation measures (SQ1–3) were 0.001–0.023, or 9700–40,500 birds of breeding age annually for the fleets included in Jiménez et al. (2020). Assuming bycatch rates in Jiménez et al. (2020) can be extrapolated to all pelagic longlining in the south Atlantic, then we estimate bycatch was 2.5–10.5 times greater than that observed by Jiménez et al. (2020), equating to 1800–7800 adults of these four species killed per year by pelagic longliners in the Atlantic south of 20°S. Full validation at the scale of the south Atlantic, of our estimates remains unattainable but it is precisely because of this issue that we adopted a risk-based approach. Without rigorous and routine observations, representative of all fleets in space and time, RFMOs should consider how such analyses can supplement best available scientific evidence in designing management measures.

To compare effectiveness of different combinations of mitigation measures independently of other factors that influence bycatch rates (Table 6), it was necessary to calculate the relative change in bycatch rates within studies between their respective trials. This implicitly assumes that these studies are broadly comparable, subject to the sources of uncertainty in Table 6. Bycatch rates in the field are strongly dependent on both the mitigation measures and abundance and composition of the seabird assemblage behind vessels. It is possible that some of the studies in our review reported lower-than-average IPUE because of low abundance of birds during their study period. However,

Table 5
ΔF (%) in estimated bycatch mortality matrix (south of 20°S). Scenario numbers: 1 = Branch Line Weighting & Night Setting; 2 = Branch Line Weighting & Bird-Scaring Lines; 3 = Bird-Scaring Lines & Night Setting; 4 = Branch Line Weighting, Night Setting & Bird-Scaring Lines; 5 = Hook Shielding Devices only.

From	To					
	SQ4	ACAP1	ACAP2	ACAP3	ACAP4	ACAP5
SQ1	−57.7	−66.5			−79.9	+49.1
SQ2	−86.1		−60.4		−93.4	−51.4
SQ3	−40.5			−15.5	−71.7	+16.5
SQ4					−52.5	+290.2

Table 6
Summary of potential error sources for bycatch mortality estimation. *Some studies have reported reduced seabird bycatch over time by individual fleets (e.g. Jiménez et al., 2020).

Potential sources of error	Bias direction
Since bycatch rates between studies cannot be further standardised, analysis assumes that bycatch rates do not vary spatially throughout the area in which the mitigation measures are applied, or over time (2012–2020)*	Uncertain
Analysis does not account for all factors relating to susceptibility per species (Jiménez et al., 2020). Differences between studies may be driven by any combination of several factors, including: • Size and composition of attending bird assemblages (i.e. if no birds are present, then IPUE will be zero, irrespective of mitigation measures applied); • competition among attending bird assemblages (e.g. size-based or between species); • diurnal-nocturnal differences in foraging between species, and influence of moon cycles or weather; • relationship between bird gape and hook size; or • influence of other individual-level vessel fishing measures (e.g. bait choice)	Uncertain
Real-world trials of some Mitigation measures (e.g. HSDs; Sullivan et al., 2017, Goad et al., 2019) outperform rSIR estimates.	Over-estimation
Some fleets do not operate year-round in the southern Atlantic (e.g. vessels targeting southern bluefin tuna that are typically not active except between March and June). Any vessel seasonality is instead expressed via the overlap of the total effort expended by fleets, both relative to one another and within each's own distribution.	Over-estimation
Scenarios assume that mitigation measures are implemented perfectly by all fleets throughout the ICCAT Convention Area	Under-estimation
Juvenile birds under-represented in tracking data but generally forage further north than adults (Gianuca et al., 2017; Carneiro et al., 2020)	Under-estimation

since rSIR is relative, this would only unduly affect the interpretation of the relative difference between conservation measure performance if there were systemic differences in the size or composition of attending bird assemblages. Although some uncertainty is unavoidable, and its true magnitude unknown, there is no empirical reason to expect that performance of any mitigation measure (individually or as a combination) is systematically biased towards higher or lower bycatch rates. Bycatch rates in the field may vary widely, but our method demonstrates that the difference in relative rates between different conservation measures can underpin substantial improvements in reducing bycatch mortality. This potential for bycatch reduction for our study populations is expected to similarly benefit all other seabird species caught on longlines in the south Atlantic (Jiménez et al., 2020).

Here, all fleets are assumed to have been fully compliant in implementing mitigation measures to either the mandated ICCAT or ACAP best practice specifications. Our estimates therefore represent a ‘best case’ scenario, rather than being more precautionary. In practice, the implementation of each combination of mitigation measures is likely to be imperfect and this will reduce its effectiveness. The performance of different combinations of mitigation measures remains a major source of uncertainty but we note that real-world data are lacking for the mitigation measures in ICCAT Rec 11–09. There is, however, empirical evidence for the measures as detailed in ACAP (2023), which draws on observations from at-sea field trials of different gears, and which found that the different combinations of mitigation measures assessed here reduce seabird mortality between 83 and 100 % relative to the controls in each study (Melvin et al., 2014; Jiménez et al., 2019b). To gain real-world data on mitigation measure performance, we encourage RFMO members to consistently and transparently report mitigation measure implementation and bycatch data.

4.2. Impacts of fishing on South Atlantic seabird populations

A key consideration in evaluating the potential impacts of fisheries bycatch on seabird populations is estimation of the levels of fishing mortality expected to cause population declines. Barbraud et al. (2008) estimated that the Crozet Islands population of white-chinned petrels would start to be ‘severely affected’ by mortality rates of 4.7 % of the population (equivalent to an instantaneous F of 0.047). However, and as with the present study, these estimations are fundamentally limited by the lack of suitable validation data to quantify the relative proportions of fishing mortality versus other sources of natural or anthropogenic mortality (Pardo et al., 2017; Dasnon et al., 2022).

4.3. Impacts on catch rates of other species

We did not systematically review potential effects of the included combinations of mitigation measures on bycatch of other species groups in the South Atlantic (i.e. turtles, sharks, or cetaceans). However, a number of the papers reviewed examined the potential impact upon catch rates of commercial species (typically tunas and billfishes) from one or more of the bycatch mitigation regimes evaluated here (Melvin et al., 2013, 2014; Sullivan et al., 2017; Debski et al., 2018; Jiménez et al., 2019b; Santos et al., 2019; Gilman et al., 2023). However, no study concluded that revising ICCAT specifications to meet ACAP (2023) guidelines significantly affect catch rates of target species.

4.4. Implementation of mitigation measures

Gilman et al. (2025) recently conducted a meta-analysis of branch line weighting regimes, specifically integrating the range of weighting specifications in RFMO regulations and ACAP best practice advice. The study demonstrated considerable variation in performance, and concluded that designs with weights of >60 g positioned >1 m from the hook reduced seabird bycatch by the greatest degree (between 89 and 93 % versus the reference case (no weighting) and specifications with lighter weights positioned closer to the hook respectively).

Kroodsma et al. (2023) estimated that globally, only 3 % of longline sets occur entirely at night, or 5.5 % in areas where night setting is required or encouraged. Setting lines early enough to finish before dawn may be difficult to consistently achieve, given the crews' other duties and hours of rest. Brothers et al. (1999) found that bycatch rates decreased by approximately 2 % for every additional 1 % of hooks that were deployed at night, up to a maximum reduction of 85 % when all hooks were deployed at night. It would therefore certainly be counter-productive to discourage operators from setting at least a proportion of hooks overnight simply because they cannot consistently set all hooks at night. We did not estimate the effect of the proportion of hooks set at night, moon-phase or weather effects, because of small sample sizes or lack of operational information.

5. Conclusions

This study is a first attempt to quantify the relative performance of currently mandated and best practice seabird mitigation measures at ocean basin scales. We conclude that:

1. The best performing set of mitigation measures was simultaneous use of all three mandated mitigation measures to ACAP best practice specifications. These three mitigation measures should be considered by ICCAT as the most appropriate management measure for seabirds until further data are available to undertake more rigorous analyses.

   Branch line weighting was the strongest individual driver of bycatch reductions between scenarios since this had the greatest difference in specifications between ICCAT and ACAP recommendations.

2. Hook-shielding devices performed similarly to other ACAP best practice guidelines for all species. Permitting the use of hook-shielding devices alongside, or as an alternative to, other mitigation measures is expected to be beneficial in reducing seabird bycatch on pelagic longline fisheries.
3. Some populations are exposed to substantial bycatch risk north of 25°S, particularly juveniles and immature birds, and best practice guidelines should be observed throughout the distribution of vulnerable species.

Data statement

The codebase and input data are supplied alongside this article. This code describes a bespoke implementation of the original EASI-Fish model framework (Griffiths et al., 2019). Interested researchers should contact Shane Griffiths (IATTC) to discuss their own applications.

CRediT authorship contribution statement

James B. Bell: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Johannes H. Fischer:** Writing – review & editing, Visualization, Software, Methodology, Investigation, Formal analysis. **Ana P.B. Carneiro:** Writing – review & editing, Methodology, Formal analysis, Data curation, Conceptualization. **Shane Griffiths:** Writing – review & editing, Software, Methodology, Formal analysis, Conceptualization. **Alessandra Bielli:** Writing – review & editing, Investigation, Data curation. **Sebastián Jiménez:** Writing – review & editing, Writing – original draft, Methodology, Data curation, Conceptualization. **Steffen Oppel:** Writing – review & editing, Methodology, Data curation, Conceptualization. **Richard A. Phillips:** Writing – review & editing, Methodology, Data curation, Conceptualization. **Helen M. Wade:** Writing – review & editing, Writing – original draft, Project administration, Methodology, Conceptualization. **Oliver Yates:** Writing – review & editing, Project administration, Methodology, Conceptualization. **Stuart A. Reeves:** Writing – review & editing, Writing – original draft, Methodology, Funding acquisition, Data curation, Conceptualization.

Declaration of Generative AI and AI-assisted technologies in the writing process

Generative AI or AI-assisted technologies were not used at any stage in the preparation of this manuscript.

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Declaration of competing interest

All authors declare that they have no competing interests or affiliations.

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Appendix A. Supplementary data

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Data availability

Shared along with manuscript files

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Glossary & Acronyms

ACAP: Agreement on the Conservation of Albatrosses and Petrels
BLW: Branch Line Weighting
BSL: Bird-Scaring Lines (a.k.a. Tori or streamer Lines)
Bycatch: Incidental capture of a non-target species during fishing operations
EASI-Fish: Ecological Assessment of the Sustainable Impacts of Fisheries (Griffiths et al., 2019)
GFW: Global Fishing Watch
GLMM: Generalised Linear Mixed Model
HSD: Hook Shielding Device
ICCAT: International Commission for the Conservation of Atlantic Tunas
IPUE: Interactions Per Unit Effort (observed). Includes birds hooked and discarded dead, release alive, and 'attacks' upon baited hooks.
RFMO: Regional Fisheries Management Organisation
rSIR: Relative Standardised Interaction Rate
UKOT: United Kingdom Overseas Territories, here referring to Tristan da Cunha, the Falkland Islands, and South Georgia and the South Sandwich Islands
WCPFC: Western and Central Pacific Fisheries Commission