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167 individuals versus millions of hooks: bycatch mitigation in longline fisheries underlies conservation of Amsterdam albatrosses

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ABSTRACT

1. Industrial fisheries represent one of the most serious threats worldwide to seabird conservation. Death of birds in fishing operations (i.e. bycatch) has especially adverse effects on populations of albatrosses, which have extremely low fecundity.

2. The single population worldwide of Amsterdam albatross (*Diomedea amsterdamensis*) comprises only 167 individuals and risks considerable decline over the mid-term from additional mortality levels potentially induced by fisheries. The priority actions listed in the current conservation plan for this species included characterizing the longline fisheries operating within its range, dynamically analysing the overlap between albatrosses and these fisheries, and providing fisheries management authorities with potential impact estimates of longline fisheries on the Amsterdam albatross.

3. During all life-cycle stages and year quarters the birds overlapped extensively with fishing effort in the southern Indian and Atlantic oceans. Fishing effort, and consequently overlap score (calculated as the product of fishing effort and time spent by the birds in a spatial unit) was highest in July–September (45% of the hooks annually deployed). Just three fleets (Taiwanese, Japanese and Spanish) contributed to >98% of the overlap scores for each stage (72% from the Taiwanese fleet alone, on average). Daily overlap scores were higher for the non-breeding versus the breeding stages (3-fold factor on average).

4. Based on previous bycatch rates for other albatross species, this study estimated that longline fisheries currently have the potential to remove ~2–16 individuals (i.e. ~5%) each year from the total Amsterdam albatross population, depending on whether bycatch mitigation measures were or were not systematically employed during the fishing operations.

5. Recent bycatch mitigation measures may be instrumental in the conservation of the Amsterdam albatross. This study suggests three further key recommendations: (1) to focus conservation efforts on the austral winter; (2) to require all operating vessels to report ring recoveries; and (3) to allocate special regulation of fishing operations in the areas of peak bycatch risk for the Amsterdam albatrosses.

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INTRODUCTION

Industrial fisheries currently constitute the major threat at sea responsible for declining seabird populations (Croxall *et al.*, 2012; Lewison *et al.*, 2012). During longline fishing operations, seabirds may be caught on baited hooks or entangled in fishing lines, resulting in their death or severe injury. It is estimated that with ~3 billion longline hooks set annually around the globe, at least 300 000 seabirds may be killed this way each year (Anderson *et al.*, 2011). Albatrosses (family Diomedeidae) are particularly vulnerable to such adverse population effects, as they are very long-lived seabirds, with low fecundity and delayed sexual maturity (Tuck *et al.*, 2001; Rolland *et al.*, 2009; Lebreton and Véran, 2013). Indeed, the worldwide decline of albatross populations occurred concomitantly with the development of industrial longline fisheries (Weimerskirch *et al.*, 1997; Tuck *et al.*, 2003), and today 17 out of 22 albatross species are listed as threatened with extinction owing to, at least to some extent, bycatch in fisheries (Croxall *et al.*, 2012; IUCN, 2012).

When Amsterdam albatrosses *Diomedea amsterdamensis* were first discovered 30 years ago in the southern Indian Ocean (on Amsterdam Island; Roux *et al.*, 1983), the small size of this sole population worldwide (just seven pairs) was attributed to mortality from longline fisheries targeting southern bluefin tuna *Thunnus maccoyii* on the birds' feeding areas in the 1960s and 1970s (Weimerskirch *et al.*, 1997). This small albatross population has since been continuously monitored with systematic individual banding, allowing its demographic structure and numbers to be accurately estimated. In 2007, the population was estimated at 167 individuals, 86 of which were mature (Rivalan *et al.*, 2010), and this increase in abundance coincided with global decreases in longline fishing effort in the southern oceans (Haward and Bergin, 2001; Tuck *et al.*, 2003; Huang and Liu, 2010).

Nevertheless, because of such low numbers the Amsterdam albatross remains sensitive to longline fisheries operating in its range (Inchausti and Weimerskirch, 2001), and is listed as 'Critically Endangered'. Indeed, if the annual additional mortality from bycatch exceeds around six individuals, the population abundance may fall below historically minimum levels within the next 50 years (with 75% probability; Rivalan *et al.*, 2010). Observations of incidental catch have not been reported for this species from any fishing fleet; however, it may occur in the high seas, outside exclusive economic zones (EEZs), where: (1) fisheries are not required to report bycatch or ring recoveries; and (2) <5% of hauled hooks are generally being surveyed by bycatch-dedicated observers onboard vessels (Huang, 2011). In fact, Taiwan (one of the world's major longline fishing countries, Huang, 2011) reported recent annual bycatch estimates of hundreds of albatrosses in the vicinity of Amsterdam Is. (218–566, from either identified or unidentified species; Huang and Liu, 2010).

In light of these threats to the long-term exsistence of the Amsterdam albatross, it is now crucial to quantify bycatch risks for the species. With that aim, the National Plan of Actions to promote the conservation of the Amsterdam albatross (2012) set up a stepwise approach. First, to improve knowledge of bird–fisheries interactions at-sea by: (1) characterizing fisheries operating within the Amsterdam albatross range; (2) analysing dynamically the overlap between albatrosses and these fisheries; and (3) evaluating utilization and risks incurred by birds in nationally- and internationally-managed areas. Second, to provide fisheries management authorities with potential impact estimates of the longline fisheries on the Amsterdam albatross (as previously conducted for other species, e.g. the shy *Thalassarche cauta* and white-capped *Thalassarche steadi* albatrosses, Baker *et al.*, 2007), and to propose adequate fishing mitigation action.

To address these aims, a substantial bird tracking database has been compiled (Thiebot *et al.*, 2014a) to analyse the spatial overlap between the albatrosses' and fishing effort distribution. It was anticipated that the number of Amsterdam albatrosses potentially at risk of incidental capture in longline fisheries varied with the birds' life-cycle stage, fishing effort (depending on season and country), and the implementation or not of bycatch mitigation measures (Tuck *et al*., 2003; Petersen *et al*., 2009; Trebilco *et al*., 2010).

MATERIALS AND METHODS

Survey of the Amsterdam albatrosses

Amsterdam albatrosses were surveyed from their sole breeding colony, on the 'Plateau des Tourbières' at Amsterdam Island (37°50′S; 77°33′E), southern Indian Ocean. Like other *Diomedea* albatrosses, this species is a biennial breeder. Rivalan *et al*. (2010) provided the following estimates of the population structure: 32 breeding, 20 failed breeding, 34 non-breeding mature, 14 juvenile and 67 immature birds for the year 2007. These estimates, together with the duration of each stage (from Roux *et al*., 1983; Thiebot *et al*., 2014a), were used to scale the relative bird occurrence at sea in each stage (Table 1). Life-cycle stages were: incubation, chick-brooding and chick-rearing (breeding stages), and the non-breeding or sabbatical phase of adults, juvenile dispersal and immaturity (non-breeding stages). Each bird studied was assigned to one of the above stages, based on season and long-term monitoring data.

The comprehensive tracking dataset detailed in Thiebot *et al*. (2014a) was used in this study (Figure 1). Three different tracking devices were used depending on the life-cycle stage (details given in Table 2). Amsterdam albatrosses are large seabirds (>6 kg), enabling tracking devices to be used with very low mass ratio (<1% in all cases). ARGOS Platform Terminal Transmitters (PTTs) were mostly used, given that these emitting devices remove the need to recapture the birds to access the location data, and have a fair spatial accuracy (from better than 150 m to 1 km). Using such units, the at-sea distribution of the birds was tracked during the incubation stage, over the whole chick-rearing stage, and during the juveniles' post-natal dispersal (over periods of months until the tags stopped transmitting). During the chick-brooding period the birds typically perform short at-sea trips that

Table 1. Duration and demographic composition of each stage within each year quarter (Q1 to Q4). The asterisk (*) indicates stages when both partners of each breeding pair are not simultaneously at sea. Demographic categories are: B: successful breeder; FB: failed breeder; NB: non-breeder; JU: juvenile; IM: immature

Stage	Year quarter	Period of each stage	Fraction of quarter	Demographic composition	Number of individuals at sea
Incubation*	Q1	28 Feb–31 Mar	0.36	B+FB	26
	Q2	1 Apr–18 May	0.53	B+FB	26
	Q3	na	0	na	0
	Q4	na	0	na	0
Chick-brooding*	Q1	na	0	na	0
	Q2	19 May–14 Jun	0.3	B	16
	Q3	na	0	na	0
	Q4	na	0	na	0
Chick-rearing	Q1	1–15 Jan	0.17	B	32
	Q2	1–15 Jun	0.18	B	32
	Q3	1 Jul–30 Sep	1	B	32
	Q4	1 Oct–31 Dec	1	B	32
Non-breeding	Q1	1 Jan–31 Mar	1	NB	34
	Q2	1 Apr–30 Jun (NB) 19 May–30 Jun (FB)	0.8	NB+FB	54
	Q3	1 Jul–30 Sep	1	NB+FB	54
	Q4	1 Oct–31 Dec	1	NB+FB	54
Juvenile	Q1	1 Jan–31 Mar	1	JU	14
	Q2	1 Apr–30 Jun	1	JU	14
	Q3	1 Jul–30 Sep	1	JU	14
	Q4	1 Oct–31 Dec	1	JU	14
Immature	Q1	1 Jan–31 Mar	1	IM	67
	Q2	1 Apr–30 Jun	1	IM	67
	Q3	1 Jul–30 Sep	1	IM	67
	Q4	1 Oct–31 Dec	1	IM	67

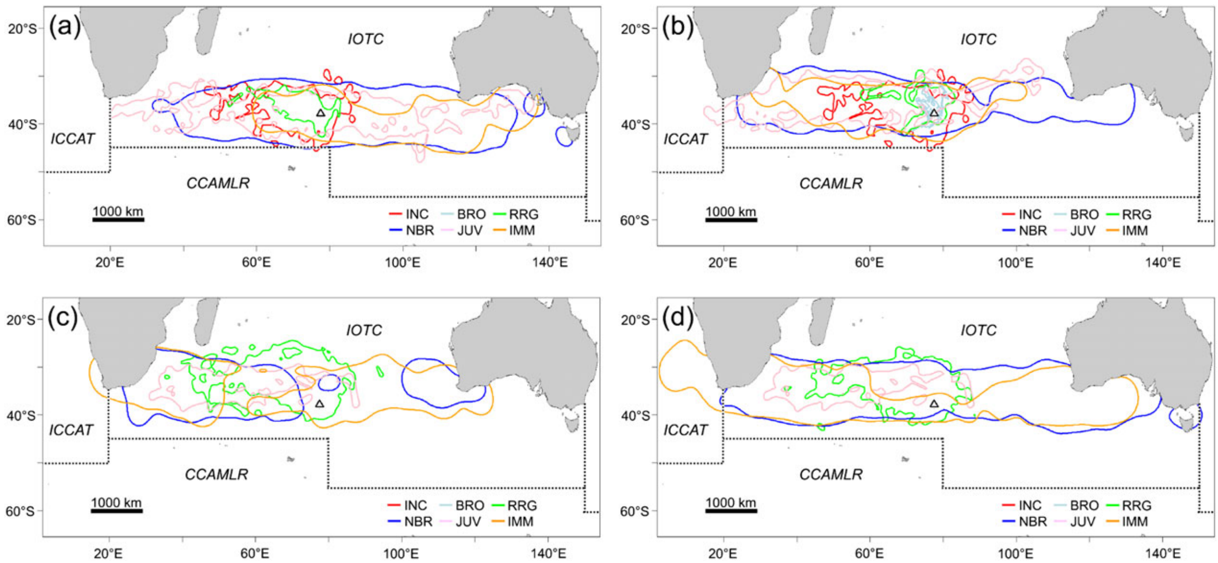

Figure 1. At-sea distribution of the Amsterdam albatrosses during the first (a), second (b), third (c) and fourth (d) year quarters, based on tracking data from Thiebot *et al*. (2014a). The lines represent the 95% kernel distribution contours calculated from the tracking data (see Thiebot *et al*. (2014a) for further details), separated by the birds' life-cycle stage (INC: incubation, BRO: brooding, RRG: chick-rearing, NBR: non-breeding, JUV: juvenile, IMM: immature). Data for incubation during the second quarter is replicated in the first quarter (no data available), and data for juvenile stage during the third quarter is replicated in the fourth quarter (see text). Boundaries of the RFMOs named in the text are indicated: ICCAT: International Commission for the Conservation of Atlantic Tunas; IOTC: Indian Ocean Tuna Commission. The adjacent area managed by the CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources) multilateral conservation agreement is also indicated. White triangle indicates location of Amsterdam Island.

Table 2. Detail of the telemetric devices deployed on the albatrosses by life-cycle stage and year: number of individuals surveyed, number of at-sea trips collected and details of the device used

Stage	Survey year	No. of individuals	No. of trips	Device used
Incubation	1996	5	7	Microwave Telemetry PTT 100 (30–50 g)
	2000	5	10	Microwave Telemetry PTT 100 (30–50 g)
	2011	14	17	Microwave Telemetry 18-g solar with duty cycle 10 h ON/24 h OFF
Brooding	2011	10	20	i- GotU GT-120 GPS (Mobile Action Technology, Taiwan, 20 g)
Chick-rearing	2011	10	269	Microwave Telemetry 18-g solar with duty cycle 10 h ON/24 h OFF
Non-breeding	2006	4	4	British Antarctic Survey (UK) MK4 GLS (5 g)
	2007	3	3	British Antarctic Survey (UK) MK4 GLS (5 g)
	2010	7	7	British Antarctic Survey (UK) MK7 GLS (4 g)
Juvenile	2005	3	3	Microwave Telemetry 50-g solar with duty cycles of 12 h ON and 24 h OFF
	2009	4	4	Microwave Telemetry 18-g solar with duty cycle 10 h ON/24 h OFF
	2012	5	4	Microwave Telemetry 18-g solar with duty cycle 10 h ON/24 h OFF
Immature	2006	1	1	British Antarctic Survey (UK) MK4 GLS (5 g)
	2011	4	4	British Antarctic Survey (UK) MK15 GLS (2.5 g)

were surveyed using the more accurate GPS loggers in 2011. GPS loggers recorded latitude and longitude with 5–10 m precision every 5 min. Finally, the adult non-breeding and the immature stages were surveyed using miniaturized Global Location Sensing loggers (GLSs). The non-breeding adult group included 12 birds in their sabbatical year (in 2006, 2007 and 2010) after successful breeding, as well as two early failed breeders in 2006. Immature birds (i.e. birds aged no more than 9 years that had never attempted breeding before) were tracked in 2006 and 2011.

Mean spatial error of location estimates is higher with GLSs (over 100 km, Phillips *et al*., 2004; Staniland *et al*., 2012) but these devices allowed the birds to be tracked during prolonged periods with minimal disturbance. GPSs and PTTs were attached to the back feathers using black Tesa tape (TM, Tesa AG, Hamburg, Germany). GLSs were mounted with cable-ties to plastic rings attached to the leg of the birds. Birds of both sexes were surveyed; sex could be assigned by comparing bill dimensions, with males having a longer and deeper bill than females, except in juvenile and immature birds (unpublished data).

Longline fishing effort data

Fish stocks, notably concerning tuna-like species, are managed throughout vast geographical areas by international organizations formed by countries with fishing interests in an area, called regional fisheries management organizations (RFMOs). Longline fishing effort (i.e. number of hooks deployed per marine sector) was based on the data reported to the RFMOs with competency for tuna-like fisheries operating within the known range of Amsterdam albatrosses. These are the Indian Ocean Tuna Commission (IOTC) and to a lesser extent International Commission for the Conservation of Atlantic Tunas (ICCAT; Figure 1). Fishing efforts in the Tasman Sea were not investigated. Indeed, the three location estimates from GLSs falling east of the IOTC boundary (longitude > 150°E) during the non-breeding stage were ignored, since these locations were only for a single individual, for less than two days (Thiebot *et al*., 2014a), and below 151°E, that is, less than the mean spatial error of GLSs (Phillips *et al*., 2004) from the 150°E boundary. Longline fishing effort datasets are freely available online (http://www.iotc.org/ and http://www.iccat.int/ for IOTC and ICCAT areas, respectively). However, for the ICCAT sector the corrected dataset described in Palma and Gallego (2009) was used. The appended corrections dealt with issues such as inconsistencies in effort units, errors in geographical coordinates, normalization of classifications, etc. Longline fishing effort data were organized on a 5°×5° spatial degree block ('spatial unit' hereafter), by year quarter (no. 1 to 4, each one comprising 3 months; i.e. quarter 1 = January, February and March, etc.) and country of origin of the operating vessels (hereafter 'fleet flag'). The average number of hooks deployed in each spatial unit per year over the last 10 years available in the databases (2000 to 2009) was used. This allowed accounting for temporal variations in the intensity and location of longline fisheries, as well as for the fact that tracking data collection spanned over a relatively long time frame. It is noteworthy that these fishing effort data may represent under-estimates (based on log-books) compared with total catch (nominal landings), and that no correction was applied here to scale up the reported effort.

Analysis of spatial overlap between distribution of birds and of fishing effort

From tracking data and fishing effort, overlap score was quantified in each spatial unit, following Cuthbert *et al*. (2005). This overlap score was defined as the product of (1) time spent by the birds, and (2) fishing effort, in each spatial unit. The tracking dataset that was used here previously demonstrated how the birds' at-sea distribution varies according to their life-cycle stage (Thiebot *et al*., 2014a). Such variations may put albatrosses at contrasting levels of bycatch risk across their life-cycle. Therefore, overlap score was quantified separately for each life-cycle stage. Moreover, the calculation of time spent by the birds in each spatial unit was compensated by the number of individuals at sea for each stage in the population (Table 1). Finally, because fishing effort data were organized into year quarters (which do not match the birds' life-cycle stages), the stages were fractioned according to the quarters. Overlap scores are given as the sum (cumulated overlap) of the values obtained in each spatial unit (Tables S1 to S6, Supplementary material); the number of spatial units where an overlap was measured is also given, as a proxy for the area over which the overlap was taking place.

Tracking data were missing for incubation during the first quarter and juvenile dispersal during the fourth quarter. To estimate the birds' distribution at sea during these periods, available tracking data for incubation during the second quarter and for juvenile dispersal during the third quarter were used. Tracking data for the closely related wandering albatross *Diomedea exulans* from the Crozet Islands, southern Indian Ocean, supported the underlying assumption that the birds' distribution was similar during the first months of incubation (Weimerskirch *et al*., 1993), as well as during the postnatal dispersal past six months at sea (Weimerskirch *et al*., 2006).

For comparison purposes between stages, the cumulated overlap scores were scaled down to daily levels by dividing the score obtained for each stage by the number of days in the corresponding stage. Comparisons were made using parametric and non-parametric tests; data normality was systematically tested using the Shapiro–Wilk test. When a significant difference was measured among more than two groups (stages, quarters or fleet flags), the Tukey's 'Honest Significant Difference' multiple comparison of means test was used to identify which group(s) differed. For all tests the significance threshold was set at $P = 0.05$. Values given are mean ± standard deviation unless specified otherwise. All overlap analyses and statistical tests were performed using R 2.15.0 (R Development Core Team, 2012).

Estimation of the number of birds at potential risk of capture at sea

A theoretical number of birds potentially captured on hooks for each stage and year quarter was extrapolated from the average number of hooks set in each spatial unit where the birds occurred, multiplied by a bycatch rate. Seabird bycatch rates per number of hooks vary extensively in the literature, depending on region, season, fishing gear and practice (Brothers, 1991; Ryan *et al*., 2002; Bugoni *et al*., 2008; Trebilco *et al*., 2010; Jiménez *et al*., 2014), but share an overall tendency to decrease over time, following the implementation of fishing practices that allow mitigating seabird bycatch at sea (Small, 2005; Gilman *et al*., 2008; Petersen *et al*., 2009; Anderson *et al*., 2011; Huang, 2011). Best-case ('0 bird caught') and worst-case (massive bird catches) scenarios of bycatch may actually occur locally, but would lead to unrealistic inferences at a larger, oceanic and year-round scale (Lewison *et al*., 2005). Consequently, two bycatch rates that seemed the most appropriate to the present study case were used. Using two different rates was expected to provide an array of realistic situations based on the fishing practice recommendations from the RFMOs for the period matching the fishing effort data (reviewed in Huang, 2011). Beginning in 2005, fishing vessels catching southern bluefin tuna were required to be equipped with two bird scaring lines (tori lines) while fishing in the three oceans in areas south of 30°S. Then, from 2008, tori lines had to be used when fishing in the Atlantic Ocean south of 20°S and in the Indian Ocean south of 28°S. The first bycatch ratio used in this study (0.0147 birds per 1000 hooks, from Huang and Liu, 2010) related to albatrosses (regardless of the species) killed by Taiwanese vessels operating in the high seas of the southern Indian Ocean and targeting southern bluefin tuna. This bycatch rate was measured over 2004–2007 and would represent a 'realistic case scenario' for the birds tracked in the present study, because of the deployment of bird-scaring lines on all fishing line sets surveyed by onboard observers. The second (0.14 albatross per 1000 hooks, from Petersen *et al*., 2009) reflected a survey spanning 1998–2005 by dedicated observers onboard Asian and South African vessels targeting swordfish *Xiphias gladius* and tuna *Thunnus* spp. off southern Africa. This figure would depict a worst-case scenario from the previous one, as during this period tori lines were deployed for only about 50% of the fishing lines set.

In order to scale the relative abundance at sea of Amsterdam albatrosses versus other albatross species likely present in the same habitat (and hence set the adjusted risk for this species to be captured among others), albatross current breeding pair numbers given by the Agreement on the Conservation for Albatrosses and Petrels (ACAP, http://www.acap.aq/index.php/en/species-assessments) were combined. Three species (the wandering, Indian yellow-nosed *Thalassarche carteri* and sooty *Phoebetria fusca* albatrosses) from six localities (Prince Edward, Crozet, Kerguelen, Heard, St Paul and Amsterdam islands) were considered the most likely occurring in the subtropical waters with the Amsterdam albatrosses. For the biennial breeding species (all but *T. carteri*), the number of annual breeding pairs given by ACAP was doubled to balance the overall number of individuals at sea regarding the annually breeding species.

RESULTS

Fishing effort

An average of 100 million hooks were set annually in the study region over the period considered. Number of hooks deployed varied significantly between year quarters (Figure 2, Kruskal–Wallis rank sum test, X^2_3=7.96, P<0.05). On average, 45% of the hooks were deployed during the third quarter, which was significantly more than during the first and fourth quarters (Tukey multiple comparisons of means, adjusted P-values =0.001 and 0.027, respectively). Three fleets contributed >97% of the average number of hooks annually deployed: these were Japan, Taiwan and Spain, in decreasing order. While Japan had the highest fishing effort of all reported fleets during the first and the fourth quarters, the Taiwanese fleet alone deployed 54% and 53% of all hooks reported in the region during the second and third quarters, respectively.

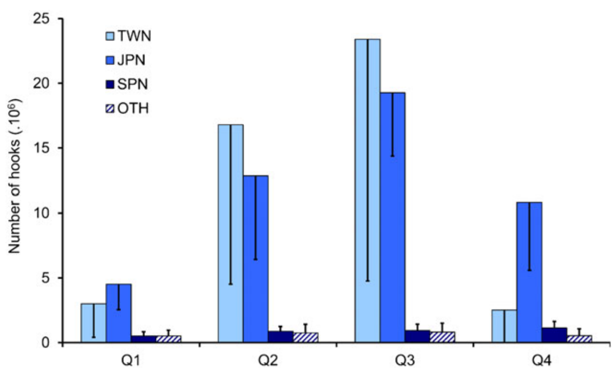

Figure 2. Total reported fishing effort (number of hooks deployed) in the distribution area of the Amsterdam albatrosses in the IOTC zone, averaged by year, for each of the four year quarters Q1 to Q4). The three fleets with the highest fishing effort (TWN: Taiwan, JPN: Japan, SPN: Spain) are distinguished; others (OTH) are pooled together on this figure.

Seasonal variations were also marked regarding the spatial distribution of fishing effort (Figure 3). In the southern Indian Ocean, peak fishing effort shifted from the west (off south-eastern Africa) to the east (off south-western Australia) over the course of the year. During the second and the third quarters, high fishing effort was found in the northern vicinity of Amsterdam Island.

Distribution overlap between albatrosses and fishing effort

During incubation and chick-brooding stages, the at-sea distribution of Amsterdam albatrosses overlapped with four different fleets, namely from Taiwan, Spain, Japan and the Seychelles (Figure 4, Tables S1 and S2). During the chick-rearing phase, cumulated overlap scores and number of fishing fleets concerned (Taiwan, Japan, Spain, the Seychelles, France, South Africa and Australia) were larger than earlier in the breeding cycle (Table S3). At-sea distribution of non-breeding adult albatrosses overlapped with 10 identified fleets (Table S4). The highest values were with the fleets of Taiwan, Japan and Spain, especially during the third quarter (Figure 5). Lower overlap scores were measured with Australian, Seychellois and other fleets. The distribution of juvenile Amsterdam albatrosses overlapped with nine distinct fleets (Table S5). Fishing efforts from Taiwan, Japan and Spain dominated the cumulated overlap scores with the birds, especially during the third quarter. Finally, during the immature phase, distribution of Amsterdam albatrosses overlapped with the highest number of fleets (12 distinct fleets, Table S6). Here again, the highest cumulated overlap scores were observed with fleets of Japan, Taiwan and Spain, especially during the third quarter.

In all, the cumulated overlap scores were highest with the Taiwanese, Japanese and Spanish fleets: these three fleets combined made up >98% of the total overlap scores (Figure 4), in line with their predominance in annual numbers of hooks set. The Taiwanese fleet alone contributed on average 72% of these cumulated values (43–96% for the immature

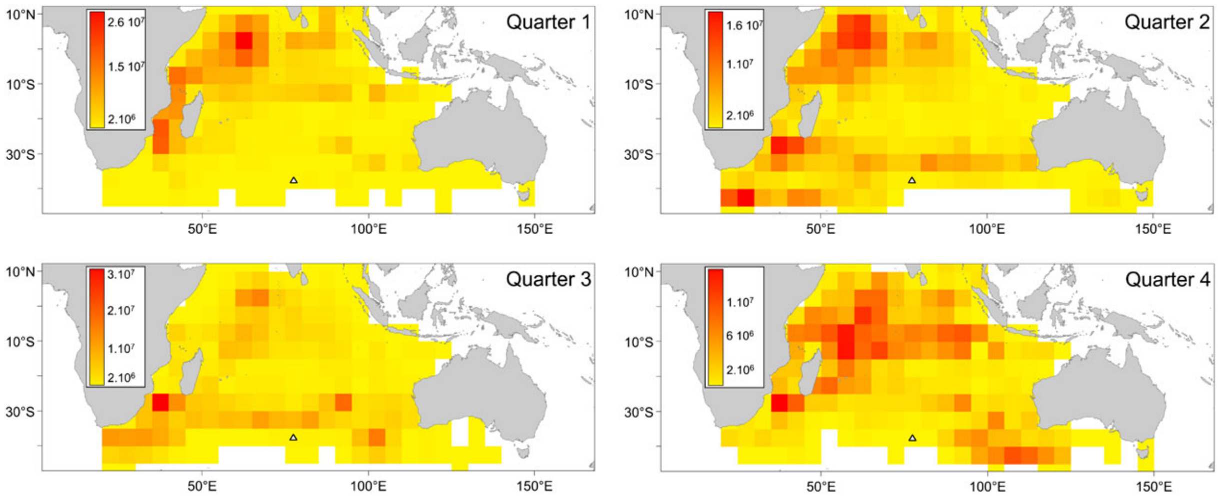

Figure 3. Total reported fishing effort (number of hooks deployed) in the IOTC zone during the period 2000–2009, for each year quarter.

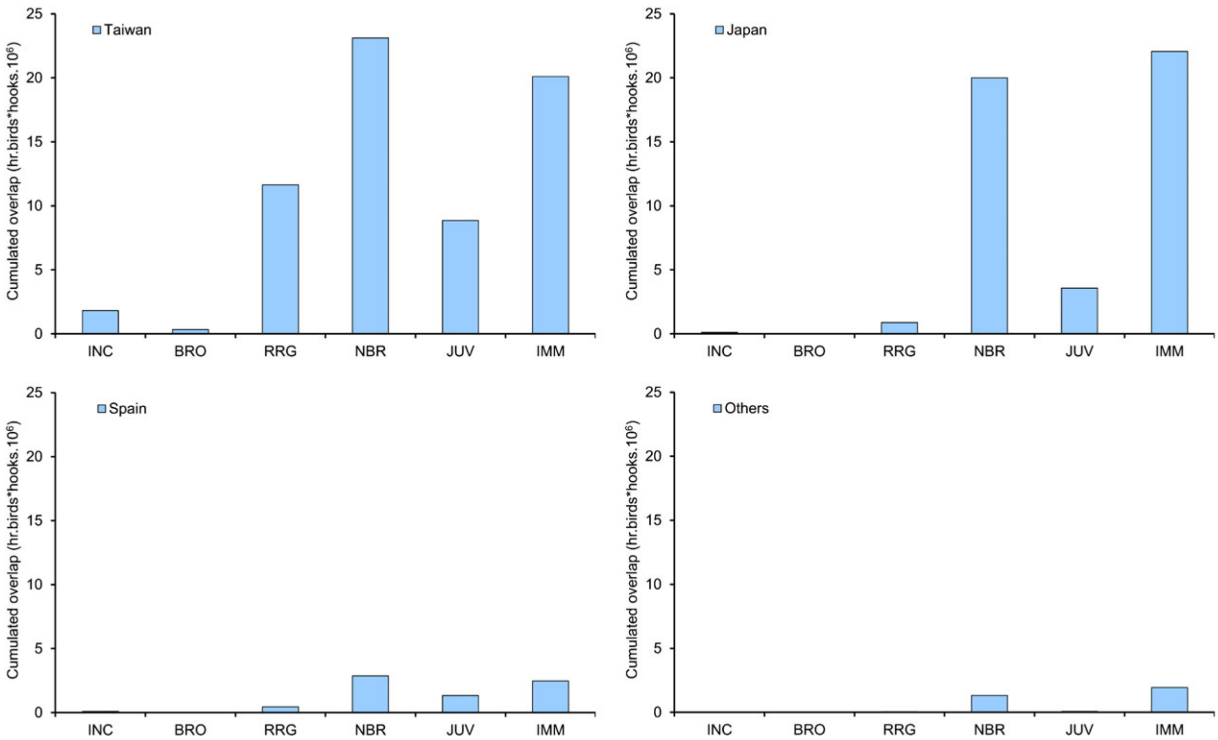

Figure 4. Score of distribution overlap between Amsterdam albatrosses and longline fishing effort at sea, separated by the birds' life-cycle stage (INC: incubation, BRO: brooding, RRG: chick-rearing, NBR: non-breeding, JUV: juvenile, IMM: immature) and the main fishing fleets: Taiwan, Japan, Spain, and undistinguished others.

and chick-brooding stages, respectively). In line with fishing effort, cumulated overlap score was systematically higher during the third quarter than during any other year quarter, for all four stages that spanned the four year quarters (Figure 5). Indeed, overlap scores for the third quarter alone contributed on average >68% of the total for the whole stage (59–87% for juvenile and chick-rearing adults, respectively; see the case given in Figure 6).

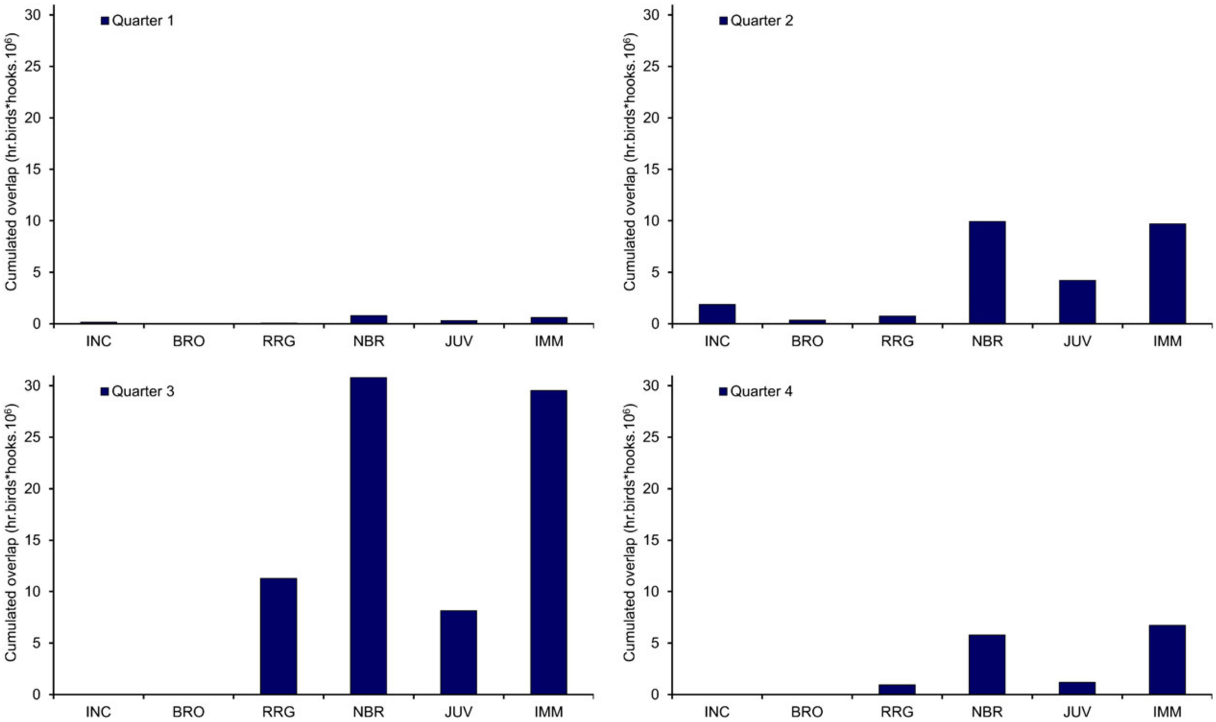

Figure 5. Score of distribution overlap between Amsterdam albatrosses and longline fishing effort at sea, separated by the birds' life-cycle stage (INC: incubation, BRO: brooding, RRG: chick-rearing, NBR: non-breeding, JUV: juvenile, IMM: immature) and the four year quarters (Quarter 1 to 4, see text).

Comparison of risk between stages

On a daily basis, the lowest overlap scores were measured during the chick-brooding stage, and the highest ones for non-breeding adults and immature birds (Table 3).

ACAP population estimates provided an occurrence rate of the Amsterdam albatross among other albatross species in the subtropical Indian Ocean, of 0.00080. Based on this occurrence rate, the summed estimates of potential bird capture in longline fisheries amounted to a total of ~2–16 birds (i.e. ~5% of the total population) per year. For six combinations of stage and year quarter, estimates of more than 1 individual potentially captured were obtained: these were for juveniles during the second quarter, for chick-rearing, non-breeding adult, juvenile and immature birds during the third quarter, and for non-breeding adults during the fourth quarter. Estimations of potential bird captures were highest for the third quarter, and that was significantly higher than for the first quarter, all stages confounded (ANOVA, F_3=3.836, P=0.0255, Tukey multiple comparisons of means:

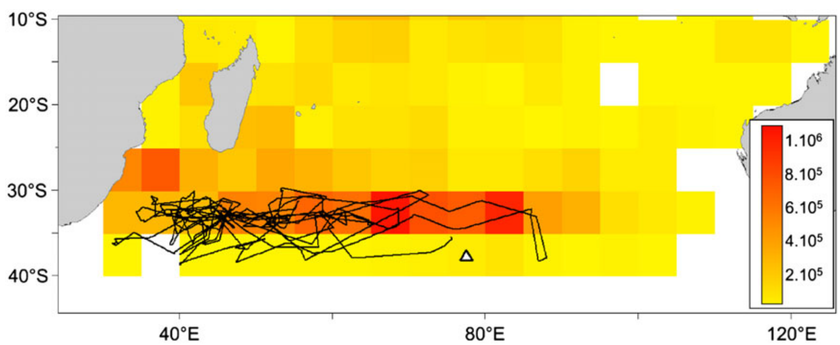

Figure 6. Example of spatial overlap between average reported longline fishing effort of the Taiwanese fleet in the southern Indian Ocean and juvenile Amsterdam albatrosses in post-fledging dispersal during the third quarter.

Table 3. Stage-specific sensitivity of the Amsterdam albatross population to bycatch in longline fisheries: estimates of average daily overlap scores between albatrosses and fishing effort at sea, and number of birds potentially captured annually on hooks, by life-cycle stage and year quarter (Q1 to Q4). Two estimates are given according to the bycatch rates and Amsterdam albatross occurrence ratio used (see text). Values >1 appear in bold. A percentage of the number of birds at potential risk of capture is given (% population) as a function of the higher estimate of birds at risk, relatively to the number of birds at this stage in the population (Table 1)

Stage	Daily overlap score (h birds. hooks^{-1})	Estimated number of birds potentially captured					% population
		Q1	Q2	Q3	Q4	Full year	
Incubation	25 608	0.0–0.0	0.0–0.3	0.0–0.0	0.0–0.0	0.0–0.4	1.5
Chick-brooding	12 926	0.0–0.0	0.0–0.0	0.0–0.0	0.0–0.0	0.0–0.0	0.0
Chick-rearing	60 559	0.0–0.0	0.0–0.1	0.2–**2.3**	0.1–0.6	0.3–**3.0**	9.4
Non-breeding	129 573	0.0–0.2	0.1–0.9	0.3–**2.6**	0.1–**1.2**	0.5–**4.9**	14.4
Juvenile	37 928	0.0–0.2	0.2–**1.6**	0.1–**1.4**	0.0–0.2	0.4–**3.4**	24.3
Immature	127 585	0.0–0.0	0.1–0.9	0.2–**2.4**	0.1–0.7	0.4–**4.1**	6.1
Total		0.1–0.5	0.4–4.0	0.9–8.6	0.3–2.7	1.7–15.8	9.3

adjusted P=0.018). Over one year, the non-breeding stages and especially the non-breeding adults accumulated the highest estimated potential risk; among the breeding stages, high estimates were obtained for chick-rearing adults only (Figure 7). Relative to the number of individuals at each stage the non-breeding stages suffered higher estimates of potential bird capture at sea (up to 24.3% of the 16 annually fledging juveniles). Among breeding stages, high ratios of individual capture were found in chick-rearing (nearly 10%).

DISCUSSION

In the framework of the current National Plan of Actions to promote conservation of the Amsterdam albatross (2012), this study highlights the following five main elements.

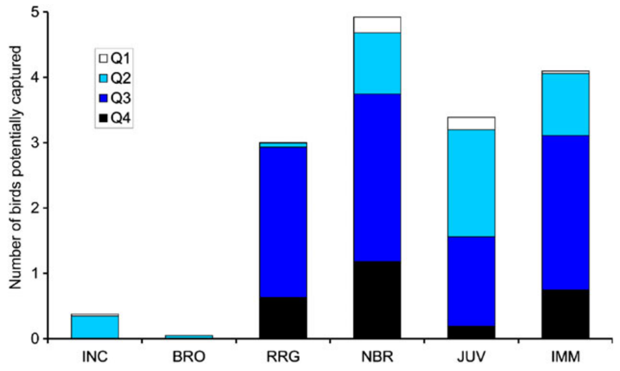

Figure 7. Relative contribution of each life-cycle stage (INC: incubation, BRO: brooding, RRG: chick-rearing, NBR: non-breeding, JUV: juvenile, IMM: immature) and year quarter (Q1 to Q4) to the total number of individuals at potential risk of capture over 1 year (considering the higher estimate of bycatch risk, see text).

First of all, this study demonstrates that the Critically Endangered Amsterdam albatross is potentially in contact with industrial longline fisheries during every stage of its life-cycle. It was previously shown that the at-sea range of this species varies dramatically depending on life-cycle stage (Thiebot *et al.*, 2014a), nevertheless this overlap with longline fisheries occurs even during the chick-brooding stage, when the birds' distribution radius is accordingly the smallest around the colony.

Second, this study provides the first figure of bycatch susceptibility for a seabird species, accounting separately for all stages. This showed a high susceptibility for the non-breeding stages, as previously suggested from seabird tracking studies across life-stages (Grémillet *et al.*, 2000; Phillips *et al.*, 2006; Trebilco *et al.*, 2008; Delord and Weimerskirch, 2011; Reid *et al.*, 2014; Thiers *et al.*, 2014), ship-based bycatch monitoring (Gales *et al.*, 1998; Petersen *et al.*, 2008) or clues found on birds (Thiebot *et al.*, 2015). These non-breeding stages usually last longer than the breeding ones (Thiebot *et al.*, 2014a), and may be the reason for the non-breeding birds accumulating bycatch risk over time. However, this study showed that even on a daily basis, overlap scores were minimal for chick-brooding, and maximal for non-breeding adult and immature birds (which may disperse across and beyond the Indian Ocean, Thiebot *et al.*, 2014a). Hence, the higher potential risk of bycatch for non-breeding Amsterdam albatrosses would stem from their greater at-sea range, exposing them to high fishing effort areas off south-eastern Africa and south-western Australia.

Third, the results clearly showed that just three fleets from the fishing effort database (from Taiwan, Japan and Spain) induced the highest overlap scores with the distribution of the albatrosses. The present study especially highlighted the risks of bycatch associated with the Taiwanese fleet alone. Indeed, this major longline fishing fleet (Huang and Liu, 2010; Huang, 2011) was unique in the IOTC region in having high fishing effort in the neighbourhood of Amsterdam Island during the second and the third quarters (Figures 3, 6 and S7).

Fourth, the third year quarter was the period with the highest distribution overlap between fishing effort and the birds. This period (July–September, the austral winter) coincided with the greatest fishing effort in the study region, possibly related to the *c*. 100 Taiwanese longliners fishing the south central Indian Ocean from June to September, mainly targeting albacore and other tunas (Tuck *et al*., 2003). Austral winter has also been highlighted as the peak season for seabird bycatch risk in the Atlantic Ocean (Tuck *et al*., 2011). However, in the Western and Central Pacific, the situation appears more complex, with higher bycatch risks in austral autumn and winter for the northern areas, but in austral summer for the southern areas (Waugh *et al*., 2012). This latter contrast demonstrates the need to account for the specificities of each fishing area (i.e. the spatio-temporal dynamics of fishing effort) to assess seabird bycatch risks in fisheries (Tuck *et al*., 2003; Baker *et al*., 2007; Trebilco *et al*., 2010; Reid *et al*., 2013; Jiménez *et al*., 2014). The results obtained in this study for the non-breeding stages, when the birds may wander at sea for more than 1 year, demonstrate the prime effect of fishing effort in producing this seasonal figure.

Finally, a total number of birds at potential risk of bycatch per year was estimated in this study. This number was either below or above the threshold of six individuals per year likely to trigger population decline (Rivalan *et al*., 2010), depending on whether tori lines were or were not systematically employed during the fishing operations (Petersen *et al*., 2009; Huang and Liu, 2010). Implementing seabird bycatch mitigation techniques during fishing operations was previously shown to significantly reduce the number of birds captured (Small, 2005), often by more than half (Petersen *et al*., 2009 and references therein). Especially, in the case of Amsterdam albatrosses, such a reduction would greatly improve the chances of persistence of the species over the mid-term, as it may lower the risks of additional mortality below the critical threshold.

Factors influencing potential bycatch estimates

In this study, an 'overlap score' was calculated, that is, a proxy of risk for a bird to be captured in each grid cell. However, this metric remains theoretical since none of the birds that were tracked in this study were actually captured. Recently, Torres *et al*. (2013) stressed the need to use fine-scale movement analysis from GPS-only data in order to reveal actual risk (i.e. not only overlap) of interaction with fishing vessels. However, such an approach is limited to the analysis of the short foraging trips for which fine-scale tracking is achievable. In the present case, all stages were surveyed, including the full adults' sabbatical year at sea, an approach requiring the use of tracking devices with low disturbance to the birds. Such devices (namely, GLSs) preclude fine-scale analysis of the birds' movements because of their current lower spatial accuracy. Moreover, the approach cannot be applied to pelagic longline fisheries because the accurate location of longline fishing vessels is not publicly available, with effort reported at a 5°×5° spatial scale and on a monthly time frame.

The number of albatrosses at potential risk of bycatch given in this study (~2–16 individuals per year) may seem under-estimated because of several factors. Specifically, the level of bycatch in the high seas is largely unknown, and it is estimated that the existence of illegal, unreported and unregulated (IUU) fisheries in the southern oceans, the non-systematic presence of onboard observers to ensure the implementation of bycatch mitigation practices, the cutting of seabirds from the line prior to observation, birds being eaten by fish prior to hauling, and birds escaping or released after capture and subsequently dying, all contribute to the potential considerable under-estimation of the total number of birds killed (Tuck *et al*., 2003). However, in view of the increasing number of breeding pairs of Amsterdam albatrosses over recent decades (Thiebot *et al*., 2014b), the extrapolated figures provided in this study are not likely to be under-estimates (at least for the higher figure) of the current situation. Two factors would support this inference. First, the higher bycatch rate used here may over-estimate Amsterdam albatross bycatch risk, since it was measured off southern Africa, where many seabirds in general and albatrosses in particular (including the Amsterdam albatross, Thiebot *et al*., 2014a) aggregate during their non-breeding period (Phillips *et al*., 2005; Petersen *et al*., 2008; Reid *et al*., 2013). Consequently, albatross bycatch rates may be higher in this region (see the very high figures given in Ryan *et al*., 2002) compared with other oceanic regions where Amsterdam albatrosses occur. Such an unbalanced situation would still allow the population to increase despite possible local bycatch, because of generally lower bycatch rates on the whole (as in other seabirds, Ryan and Ronconi, 2011). Second, bycatch rates are expected to be lower during recent years in the distribution range of the Amsterdam albatross. This is due to the increase in awareness and actions taken against seabird bycatch by the RFMOs, and implemented by the vessels (Huang, 2011). Important progress made in this context that may benefit Amsterdam albatrosses has been the recent extension of the area where operating vessels are required to implement bycatch mitigation measures, shortly followed by the obligation for these vessels to implement at least two kinds of the recommended mitigation measures simultaneously (formerly at least one; Huang, 2011; IOTC, 2011, 2012) in the regulated areas. These ‘best fishing practices’, encompass night-setting of the fishing lines (Bull, 2007; ACAP, 2013; RSPB, 2013), a measure that may dramatically reduce the incidental capture of albatrosses in particular, as they are mainly active during the day only (Cherel *et al*., 1996; Weimerskirch *et al*., 2000; Jiménez *et al*., 2014).

Key recommendations for conservation of the Amsterdam albatrosses at sea

Thiebot *et al*. (2014a) previously clarified the at-sea distribution range of Amsterdam albatrosses and their utilization of nationally- versus internationally-managed areas. With the present study, new actions for improving knowledge of bird–fisheries interactions at-sea were fulfilled. Further goals listed in the framework of the Amsterdam albatross conservation plan were, first, to propose adequate fishing action layouts. While some RFMOs have a purely advisory role, most (such as IOTC and ICCAT) have management powers to set catch and fishing effort limits, technical measures, and control obligations beneficial to seabird conservation (Small, 2005; IOTC, 2011, 2012). In this context, the following further recommendations can be proposed to help with the conservation of Amsterdam albatrosses in both ICCAT and IOTC regulated areas.

First of all, higher fishing efforts in the distribution range of the Amsterdam albatross coincided with the austral winter, leading to the greatest bycatch risks for the species at that time. While reducing fishing efforts in winter would obviously be an option to lower seasonal bycatch risks, another option would be to optimize the use of yearly available human and financial resources in the ongoing efforts made to develop bycatch monitoring programmes at-sea (Huang, 2011). For the Amsterdam albatross, increasing the coverage of fishing operations in the distribution range of the species by dedicated observers during the austral winter would be the most efficient use of resources. This should result in further reducing the bycatch figures occurring on these vessels, by ascertaining the actual and successful implementation of bycatch mitigation measures at sea. Indeed, the unsuccessful deployment of a scaring device may cause increased bycatch figures (Weimerskirch *et al*., 2000).

A second improvement would be for RFMOs to require all operating vessels to report ring recoveries. This simple measure, though it would not prevent bycatch, would allow researchers to considerably refine population-specific bycatch patterns in demographic surveys. This would be especially significant in the case of Amsterdam albatross captures, since all birds are leg-banded (Rivalan *et al.*, 2010). In addition, reporting ring recoveries would allow a much better post-mortem identification of the birds. Again, this would be extremely beneficial in the specific case of the Amsterdam albatross as these birds may easily be mistaken for immature specimens of other great albatross species (Roux *et al.*, 1983). This would be the only way to refine the risk for Amsterdam albatrosses of interacting with fishing vessels at sea, in comparison with other albatross species. Indeed, taking into account any specific degree of attraction to or avoidance of fishing vessels (Hyrenbach, 2001; Granadeiro *et al.*, 2011; Barbraud *et al.*, 2013) or interspecific associations (Jiménez *et al.*, 2012) could considerably refine the estimates of bycatch for one particular species.

Third, the very high overlap scores found between the Taiwanese fleet and the Amsterdam albatrosses during the second and third quarters reflect the fact that no other fleet had similarly high fishing effort in the immediate vicinity of Amsterdam Island (Figures 3, 6 and S7). This particular area of peak bycatch risk for the Amsterdam albatrosses may thus deserve special regulation on the limitation of fishing effort, or exceptionally high coverage of fishing operations by dedicated observers. In addition, two other areas (off south-eastern Africa and south-western Australia) also appeared to be hotspots of potential albatross bycatch, and these may be used by other species as well (Baker *et al.*, 2007; Reid *et al.*, 2013), thus increasing the benefit of a special protection status of the area to other species.

In conclusion, this study showed that risks of bycatch for the Amsterdam albatross are unevenly distributed across population categories, fleet flags and seasons. Risks would be higher for, respectively, the non-breeding categories, three fleets (the Taiwanese, Japanese and Spanish) and during the austral winter. The extrapolated potential bycatch estimates provided in this study (~2–16 individuals per year) suggest that the recent bycatch mitigation measures may be instrumental in the conservation of the single population of Amsterdam albatrosses worldwide.

The recent regulations on seabird bycatch mitigation may have, in turn, affected fishing effort and distribution, as previously observed in the context of changed regulations (Haward and Bergin, 2000, 2001; Tuck *et al.*, 2003; Huang, 2011; Ting *et al.*, 2012). This suggests that an update of the overlap analysis may be run again when new bycatch estimates become available in this region (reflecting the implementation of the most recent regulations), together with the up-to-date fishing effort dataset. Undeniably, the RFMOs' ongoing effort to gather and correct fishing effort data from several countries (such as Indonesia and the Islamic Republic of Iran in the IOTC region; IOTC, 2015) may reveal new challenges for seabird conservation in the near future.

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