



HAL
open science

Fine-scale interactions between boats and large albatrosses indicate variable susceptibility to bycatch risk according to species and populations

A. Corbeau, J. Collet, F. Orgeret, P. Pistorius, H. Weimerskirch

► **To cite this version:**

A. Corbeau, J. Collet, F. Orgeret, P. Pistorius, H. Weimerskirch. Fine-scale interactions between boats and large albatrosses indicate variable susceptibility to bycatch risk according to species and populations. *Animal Conservation*, 2021, 24 (4), 10.1111/acv.12676 . hal-03138918

HAL Id: hal-03138918

<https://hal.science/hal-03138918v1>

Submitted on 28 Oct 2021

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

1 **Fine-scale interactions between boats and large albatrosses**
2 **indicate variable susceptibility to bycatch risk according to species**
3 **and populations**

4

5 Alexandre CORBEAU^{1*}

6 Julien COLLET²

7 Florian ORGERET^{3,4}

8 Pierre PISTORIUS^{3,4}

9 Henri WEIMERSKIRCH¹

10

11 ¹Centre d'Études Biologiques de Chizé, UMR7372 CNRS-La Rochelle Université, 79360
12 Villiers en Bois, France.

13 ²University of Oxford, Department of Zoology, Oxford OX1 2JD, United Kingdom.

14 ³DST/NRF Centre of Excellence at the FitzPatrick Institute for African Ornithology,
15 Department of Zoology, Nelson Mandela University, Port Elizabeth 6031, South Africa.

16 ⁴Marine Apex Predator Research Unit, Institute for Coastal and Marine Research, Nelson
17 Mandela University, Port Elizabeth 6031, South Africa.

18

19

20

21 Address for correspondence: corbeau.alexandre@gmail.com

22

23 **Abstract**

24 Many seabirds are attracted to fishing boats where they exploit foraging opportunities which
25 often involve bycatch-related mortality. Bycatch risk is generally estimated by overlapping
26 seabirds foraging ranges with coarse-scale monthly maps of fishing efforts. A more direct
27 estimation would be the time birds spend attending fishing boats.

28 Here we matched data from Automatic Identification Systems from all declared boats in the
29 Southern Ocean, with 143 simultaneous foraging trips from all populations of large
30 albatrosses (*Diomedea amsterdamensis* and *Diomedea exulans*) breeding in the Indian Ocean
31 (Marion, Crozet, Kerguelen, Amsterdam islands). We quantified and compared real-time co-
32 occurrence between boats and albatrosses, at different scales (100, 30 and 5km). We also
33 examined to what extent co-occurrence at a large-scale (5x5° grid cell) predicted fine-scale
34 attendance (5km).

35 Albatrosses on average spent about 3h per trip attending fishing boats (<5km) at both
36 Amsterdam and Marion and about 30h per trip at Kerguelen. In all populations >90% of
37 declared fishing boat attendance occurred within Economic Exclusive Zones (EEZ) where
38 bycatch mitigation measures are more strictly enforced. Outside EEZs, birds from all
39 populations also significantly attended non-fishing boats. Fishing boat density at large scales
40 (5x5°, 100km) poorly predicted time spent attending fishing boats (<5km) across populations.

41 Our results indicate a large variation in fishing boat density within foraging ranges of
42 different populations, and in time spent attending boats. We discuss the pros and cons of using
43 large-scale analyses and how they might be improved to better estimate bycatch risks in
44 seabirds when fine-scale data is available particularly for conservation purpose on those
45 highly threatened species.

46 **Key words:** albatross populations; biologging; bycatch assessment; ecological trap; fine-scale
47 interaction; fisheries.

48 **Introduction**

49 In marine ecosystems, together with climate change, industrial fisheries constitute the main
50 driver of ecological deterioration (Pauly et al., 2002). Fisheries interact with marine predators
51 mainly by competing for resources (Cury et al., 2011; Grémillet et al., 2018) and by inducing
52 mortality through bycatch of non-target species (Lewison et al., 2004). Fisheries can also
53 facilitate access to prey for higher predators or provide additional food resources (Oro et al.,
54 2013). For all these reasons many species of seabirds and marine mammals are attracted to
55 fishing boats (Votier et al., 2004; Read, 2008; Brothers et al., 2010; Bugoni, McGill, &
56 Furness, 2010) in search of foraging opportunities associated with fishing bait or discards
57 (Votier et al., 2004; Bicknell et al., 2013a). However, the associated bycatch is one of the
58 primary threats for seabird populations around the world (Croxall et al., 2012). Moreover in
59 some seabird populations the poor quality of these food resources negatively affect
60 reproductive success (Gremillet et al., 2008; Le Bot, Lescroël, & Grémillet, 2018a). Another
61 concern is that populations heavily reliant on fishing vessels for food resources may be
62 negatively impacted by changes in fishing policies (Bicknell et al., 2013b).

63 Bycatch is the most important threat for albatrosses and large petrels while at sea with
64 high levels of mortality often induced by long-line fisheries (Delord et al., 2005; Anderson et
65 al., 2011; Croxall et al., 2012). In the Southern Ocean, albatrosses overlap extensively with
66 long-line fisheries, targeting tuna in oceanic waters, and various species of bottom-dwelling
67 fishes over shelves and shelf-edges, in international waters as well as the Economic Exclusive
68 Zones (EEZ) of the respective countries. The extent of spatio-temporal overlap between
69 different fisheries and albatross foraging grounds has been inferred to represent mortality risk

70 for various populations (Bertrand et al., 2012; Clay et al., 2019; Heerah et al., 2019).
71 However, information on fisheries location is generally available at a large scale, especially in
72 international waters. For example, global fishing efforts provided by Regional Fisheries
73 Management Organisations for tuna and billfishes is only available at a monthly and by 5x5°
74 cell resolution (Clay et al., 2019; Heerah et al., 2019). This approach ignores the possibility
75 that fisheries and seabirds could co-occur at a large scale without birds interacting with the
76 fishing boats, particularly if they are not attracted by vessels (Clark et al., 2020). To better
77 estimate mortality risk it is therefore necessary to complement these approaches with more
78 direct information on the actual time birds spend attending fishing boats and how this varies
79 spatially (Torres et al., 2013). This has been hampered in the past by difficulties in obtaining
80 fine scale information on fishing vessel movements from fishing operators or authorities, with
81 Vessel Monitoring System (VMS) information often being confidential (Votier et al., 2010)
82 and restricted to confined EEZ territories.

83 In the Indian Ocean, large active longline tuna and Patagonian toothfish (*Dissostichus*
84 *eleginoides*) fisheries (Delord et al., 2005) overlap with the foraging ranges of the two large
85 albatross species (wandering - *Diomedea exulans* and Amsterdam - *Diomedea*
86 *amsterdamensis*) breeding in the region (Henri Weimerskirch, Brothers, & Jouventin, 1997;
87 Delord et al., 2005). The past decline of the former species has been attributed to bycatch
88 associated with long-line fisheries (Brothers, 1991; Henri Weimerskirch, Brothers, &
89 Jouventin, 1997; Nel et al., 2002). Despite mitigation measures that have been implemented
90 by toothfish longline fisheries within the EEZs, which has resulted in a reduction in bycatch
91 by this fishery (Delord et al., 2005; Henri Weimerskirch et al., 2018), there are still concerns
92 of bycatch risk from long line fisheries and/or illegal or uncontrolled fisheries targeting tuna
93 in international waters within this region (Brothers, 1991; Henri Weimerskirch, Brothers, &
94 Jouventin, 1997; H. Weimerskirch et al., 2020). Moreover, within more regulated EEZ waters

95 it is important to estimate the extent to which albatrosses of different species and populations
96 spend interacting with toothfish long-liners, to better quantify potential sub-lethal issues of
97 dependence and possibly poor foraging quality (Bicknell et al., 2013a; Le Bot, Lescroël, &
98 Grémillet, 2018b).

99 In this study, we combined a large tracking data set of foraging albatrosses (H.
100 Weimerskirch et al., 2020) with the locations and types of all declared boats from Automatic
101 Identification System (AIS) in the southern Indian Ocean. GPS tracking data were collected in
102 2018/2019 on breeding adults from all four major populations of large albatrosses in the
103 Indian Ocean. We spatio-temporally matched these datasets to estimate the degree of co-
104 occurrence at various scales from seascape (<100km), through encounter (<30km) to
105 attendance (<5km), following Weimerskirch et al. (2020). We particularly focused on the
106 time spent attending fishing boats (<5km) as a potential proxy for bycatch and other boat-
107 associated risks. We examined how it differed among individuals and populations, how it
108 differed with different types of fishing and non-fishing boats present during the breeding
109 season, and how it differed between EEZs around subantarctic islands and international
110 waters where different fisheries operate with different mitigation measures. Finally, to assess
111 to what extent co-occurrence at a larger-scale reflects co-occurrence at finer-scale and could
112 be used as a proxy for bycatch risk, we compared the time spent attending fishing boats
113 (<5km) to the encounter rate (30km) and the density of boats in the seascape (<100km) as
114 well as to another, more widely used method of aggregating boat data by Regional Fisheries
115 Management Organisations (RFMO) 5x5° grid. We 1) hypothesized that there is a large
116 variation in the levels of exposure to boats according to albatrosses' foraging zones and range,
117 2) tested to what extent it resulted in variation in the time spent attending boats and 3) tested
118 whether large scale 5x5° grid methods provide an adequate reflection of the attendance to

119 boats and therefore the risk of bycatch. We then discuss implications for bycatch and sub-
120 lethal risks to the different populations.

121

122 **Materials and methods**

123 Field sites

124 Fieldwork was carried out in French southern territory (Crozet, Kerguelen & Amsterdam)
125 during the course of a large-scale Ocean Sentinel program between January and April within
126 the 2018/2019 breeding season of large albatrosses in the Southern Indian Ocean (H.
127 Weimerskirch et al., 2020). We deployed loggers on wandering albatrosses at Possession
128 Island (Crozet Islands) and at the Kerguelen Islands, and on Amsterdam albatrosses at
129 Amsterdam Island. During the same incubation season GPS loggers were deployed on
130 wandering albatrosses on South Africa's Marion Island, Prince Edward Islands.

131 Loggers

132 On Crozet, Kerguelen and Amsterdam, Centurion loggers (65g) recording GPS location every
133 2 min were deployed on incubating birds for one or two successive foraging trips (H.
134 Weimerskirch et al., 2020). On Marion, GPS loggers (IgotU, 60g) recording locations every
135 20 min were deployed for several trips during the incubation and brooding period. The
136 loggers were attached to the back feathers with Tesa® Tape (Germany), and represented
137 between 0.5 and 0.85% of large albatross body mass, much less than the maximum 3%
138 recommended for loggers attached on flying seabirds (Phillips, Xavier, & Croxall, 2003).

139 GPS data and AIS Dataset

140 A total of 143 trips on incubating albatrosses was recorded, with 57 trips from 27 individuals
141 on Marion Island, 10 trips from 8 individuals at Amsterdam, 49 trips from 49 individuals at
142 Crozet and 27 trips from 24 individuals at Kerguelen.

143 After using speed filters (150 km.h⁻¹; Weimerskirch et al., 2020), we divided tracks
144 by trips (removing location on land) and we only used trips from the incubation period.

145 AIS data for all fishing and non-fishing boats (Fig. S1) were obtained from French
146 satellite transmission society (*Collecte Localisation Satellites*) for the study period for the
147 sector 10°-180°E, 20°-70°S through the Ocean sentinel program (H. Weimerskirch et al.,
148 2020), providing a total of 120 million AIS locations. Through the AIS system, in addition to
149 regular GPS locations (mean resolution of 10 min) we obtained continuous data on
150 identification name, nationality, type of boat (fishing or not), and activity for all declared
151 boats in the Southern Indian Ocean. AIS data and bird locations were spatio-temporally
152 matched following Weimerskirch et al. (2020) to produce a dataset where all GPS locations of
153 each bird from each population are associated to the presence/absence, number and types of
154 boats transmitting AIS information within ranges of 100, 30 and 5 km from birds.

155 These different radius distances from birds are used to characterize the ‘boat seascape’
156 (within 100 kilometres around the tracked bird), the ‘boats encountered’ (30km) and the
157 ‘boats attended’ (5km). The 30 km distance was used as it is the distance within which an
158 albatross can visually detect a boat (Thiebault et al., 2014; J Collet, Patrick, & Weimerskirch,
159 2015; Pirota et al., 2018). The 5km threshold is close to the distance at which wandering
160 albatrosses are seen to engage in specific foraging behaviour around boats (3km; Collet et al.,
161 2015) and is used to facilitate comparisons with previous studies that used radar detectors
162 with a range detection of 5km (Weimerskirch et al., 2020, 2017).

163 We defined ‘events’ (attendance and encounter events) as periods of consecutive bird
164 locations within the respective distances of at least one boat with time gap less than 2 hours.
165 To compare sites, and accommodate the relatively coarse scale GPS sampling at Marion
166 Island, we removed all events (attendance and encounter) lasting less than 20 min. This
167 procedure also limits the effects of uncertainties on “instantaneous” bird-boat distances (H.
168 Weimerskirch et al., 2020). We also removed the few incomplete trips for presenting trip
169 statistics (Table 1).

170 To compare with other studies using large scales 5x5° of fishing effort provided by
171 RFMOs (Clay et al., 2019; Heerah et al., 2019), we merged all AIS locations present during
172 the study period within grid cells of 5x5° (Fig. S1).

173 Environmental variables

174 AIS data do not provide detailed information on the type of fishing gears nor the mitigation
175 measures employed by fishing boats. We tried to further infer these information from the
176 waters they operated in. We added bathymetry data to each bird location (R package
177 ‘marmap’, Pante and Simon-Bouhet, 2013), which was extracted from ‘ETOPO1 Global
178 Relief Model’ from ‘National Oceanic and Atmospheric). We used it to categorize bird
179 locations as on a shelf or a shelf-edge (above -2000 m), where mainly benthic fishes are
180 targeted, or not, where tunas are the main target. We also considered whether locations were
181 within EEZ or not (data from <http://www.marineregions.org>) and separately considered the
182 time in attendance for specific EEZs with enforced mitigation measures within the range of
183 our populations (Crozet, Kerguelen, Heard, McDonald Saint-Paul and Amsterdam Islands).
184 Finally, from estimates of the locations of the polar front (Moore, Abbott, & Richman, 1999)
185 and the subtropical front (Belkin & Gordon, 1996) we further categorized bird locations into
186 Antarctic, subantarctic and subtropical waters.

187 Analyses

188 For visualization purposes, we used kernel Utilization Distributions (UDs 50 and 90%), using
189 the R package ‘adehabitatHR’ (smoothing parameters, h=1 degree).

190 To compare different parameters (Table 1) between each population, we used linear
191 mixed model or generalized linear mixed models (depending on the distribution, using R
192 package “fitdistrPlus”). Negative binomial family were used for over-dispersed count data
193 and binomial family for ratio data (R packages ‘lme4’ and “lmerTest”). Bird individual
194 identities were used as random factors. We further used post-hoc tests (Tukey tests, R
195 package multcomp) and Holm-Bonferroni correction for P values. We used Chi² test to
196 compare distributions of the number of trips with or without boat interaction between
197 populations.

198 To compare AIS data to time birds spent per 5x5° grid, we summed AIS locations (for
199 all types of boat or only for fishing boat) per grid cell used by each study population during
200 their respective incubation-period months (April for Amsterdam birds, January and February
201 for Crozet and Kerguelen birds, and February and March for Marion birds). Then, we used
202 Pearson correlation to examine whether those aggregated AIS data are related to the time
203 spent by birds in general, with boats in their seascape (<100km), with boats encountered
204 (<30km) and with boats attended (<5km) in the same 5x5° grid cells.

205

206 **Results**

207 For the 143 trips recorded during incubation, there were no significant differences between
208 populations in the duration of foraging trips but mean maximum distance from the colony
209 differed between Kerguelen (shortest) and Marion (longest) (Table 1).

210 Birds from Kerguelen spent more time foraging within EEZs ($74\% \pm 32$, Table 1) than
211 birds from Crozet ($57\% \pm 35$), Amsterdam ($39\% \pm 41$) and Marion ($36\% \pm 24$). Amsterdam and
212 Marion birds spent less time foraging over shelf waters ($20\% \pm 29$ and $11\% \pm 15$ respectively)
213 as compared to Crozet ($40\% \pm 33$) and Kerguelen birds ($65\% \pm 29$) (Fig. 1) (Table 1 and Table
214 S1 for test values).

215 Amsterdam albatrosses spent most of their time in subtropical waters ($97\% \pm 07$)
216 whereas the three wandering albatross populations foraged mainly in subantarctic waters
217 ($57\% \pm 31$ to $78\% \pm 30$) (Fig. 1) (Table 1 and Table S1 for test values).

218 Among the 143 trips recorded, the percentages of trips with at least one boat within
219 100 km (boat seascape), were significantly different between populations, ranging from 68%
220 to 100% ($\text{Chi}^2, 3 = 24.9$; $p \text{ value} = 1.5e-05$) (Table 2). The percentage of trips with boats
221 encountered (within 30 km) also varied significantly between sites, from 63 to 85% ($\text{Chi}^2, 3 =$
222 9.08 ; $p \text{ value} = 0.028$) (Table 2). Finally, the percentage of trips with attendance (within 5
223 km) of boats were also significantly different between sites, varying from 47 to 73 % ($\text{Chi}^2, 3$
224 $= 8.01$; $p \text{ value} = 0.046$) (Table 2).

225 The number of encounters and attendance events per trip, when considering fishing
226 boats and other boats together (transport, tankers, etc.) was similar between populations
227 (Table 1 and Table S1 for test values). Kerguelen birds spent more time on average per trip
228 within 30km of all types of boats ($53h \pm 62$), within 5km of all type of boats ($31h \pm 38$) and
229 within 5 km of fishing boats ($30.6h \pm 39$) (Fig. 2a) than birds from other populations (Table 1
230 and Table S1 for test values). Similarly, Kerguelen birds spent significantly more time
231 attending boats inside EEZs (with mitigation measures) than birds from other populations.
232 However, outside EEZ (where mitigation measures are less controlled) the different

233 populations spent similar time attending AIS-recorded boats of all types, and similar time
234 attending AIS-recorded fishing boats (Table 1 and Table S1 for test values).

235 Based on all location, Amsterdam birds on average had the most number of boats
236 (1.9 ± 5) and the most number of fishing boats (0.83 ± 1.9) in their seascapes ($<100\text{km}$), at least
237 twice as much as other populations (Table 1 and Table S1 for test values). However at
238 Kerguelen birds on average had the most number of boat encounters ($<30\text{km}$: 0.2 ± 0.4), the
239 most number of boats attended ($<5\text{km}$: 0.1 ± 0.3 , Fig. 2b) and the highest ratio of the number
240 of boats attended relative to the number boats in the seascape (0.3 ± 0.5), most of the time by a
241 factor of 5-10 fold compared to other populations (Table 1 and Table S1 for test values).

242 Marion and Amsterdam birds had a smaller proportion of fishing versus non-fishing
243 boats in their seascapes ($<100\text{km}$) compared to other populations (Table 1 and Table S1 for
244 test values). Marion birds had a significantly lower proportion of fishing boats among
245 encountered boats ($<30\text{km}$) than Crozet and Kerguelen and slightly less than Amsterdam. The
246 proportion of fishing boats among attended boats ($<5\text{km}$) was not different between Marion,
247 Amsterdam and Crozet birds (0.22 ± 0.4 , 0.64 ± 0.4 and 0.76 ± 0.4 , respectively), but it was lower
248 than for Kerguelen birds (0.95 ± 0.2 , Table 1 and Table S1 for test values).

249 Finally, we found that at all locations, birds attended only a small proportion of the
250 total number of boats in their seascapes: $30\% \pm 45$ for Kerguelen birds which was significantly
251 higher than for Crozet birds ($7\% \pm 24$) and for Marion and Amsterdam birds ($4\% \pm 0.19$;
252 Table 1 and Table S1 for test values).

253 For all four populations, there were no significant correlations between the time spent
254 by birds per $5 \times 5^\circ$ grid cells (in general, with boats in their seascape, with boats encountered
255 and with boats attended) and between the number of AIS signals per $5 \times 5^\circ$ grid cells. This

256 applied when considering all types of boats as well as fishing boats only (Table 3 and Fig.
257 S1).

258

259 **Discussion**

260 Our study clearly indicates strong differences between populations in the time spent attending
261 boats, with different associated bycatch risks (Fig. 2a). Furthermore, we clearly show that these
262 variations in time spent attending boats are not a simple linear function of the density of boats
263 in the seascape, as previous methods aimed at assessing bycatch risk have assumed. Indeed, we
264 have shown that using AIS data combined with fine scale GPS tracking of seabirds can provide
265 a considerably more reliable estimate of bycatch risks, through the documentation of the actual
266 time birds spend interacting at a fine scale with different types of declared boats. Indeed, most
267 previous studies used monthly maps of the number of hooks deployed within aggregated $5 \times 5^\circ$
268 cells (around 560×560 km in our region) to estimate risks incurred by foraging birds (Clay et
269 al., 2019; Heerah et al., 2019), but here we show that analyses at this scale do not correlate at
270 all with time spent interacting with boats.

271 Overall, we found that all four populations spent considerably more time attending
272 fisheries boats within EEZs (with bycatch mitigation measures) than in international waters. On
273 average, birds from all populations spent less than 1h per trip attending declared fishing boats
274 outside EEZs where bycatch mitigation measures are generally adopted leading to low seabirds'
275 mortalities. While it means that at least 3 out of the 4 studied populations are indeed at risk of
276 bycatch from these declared fleets outside EEZs (where no bycatch mitigation measures are
277 taken), birds seem to spend limited time attending them. Important to note is that this result
278 could potentially be very different if non-declared boats (without AIS) could also be included.
279 Indeed, illegal, undeclared and unregulated fleets may represent up to 30% of boat encounters

280 for breeding large albatrosses (H. Weimerskirch et al., 2020). The lack of information on these
281 boats can partly be remedied by using new loggers that can detect radar emission of boats up to
282 5km away (Weimerskirch, Filippi, Collet, Waugh, & Patrick, 2017). However, AIS data
283 provides additional information on boat identity and also on a larger scale boat seascape around
284 birds, so that more accurate results could be reached by combining the two methods (radar
285 detectors were not available for Marion birds in the present study).

286 AIS data revealed that the four populations of large albatrosses breeding within the
287 Indian Ocean foraged in very different boat seascapes. Yet, the time spent interacting with
288 fishing boats within or beyond EEZs could not be predicted by the respective boat densities
289 with the foraging ranges of the different populations. Fishing boats in the area mainly fall within
290 two categories: toothfish longline fisheries operating on subantarctic shelves and tuna
291 longlining fisheries operating in subtropical waters (Corbeau et al., 2019). In addition, many
292 non-fishing boats (cargo, tankers) transit through subtropical waters between Africa, Asia and
293 Australia. Birds from Marion had the least contact with boats, with a large proportion of trips
294 without boat encounters, yet most attended boats were not associated with fisheries. This can
295 potentially be explained by the absence of a surrounding shelf and the small scale of the
296 declared toothfish fishery in the region in comparison to Crozet and Kerguelen (H.
297 Weimerskirch et al., 2020). Furthermore, birds from Marion spent a relatively low proportion
298 of their foraging time in subtropical waters (in contrast to Amsterdam birds). At the opposite
299 end, Amsterdam birds ranged in the densest boat seascape, both in terms of the general boat
300 density and in terms of fishing boats density. This is of particular concern for this endangered
301 species with less than 60 pairs breeding annually on Amsterdam Island (Thiebot et al., 2015;
302 Heerah et al., 2019). Yet, birds from Marion and Amsterdam populations eventually spent very
303 similar average amounts of time with declared fishing boats both within or outside EEZs (with
304 different bycatch mitigation measures). In contrast, compared to Marion and Amsterdam

305 populations, Kerguelen birds spent considerably more time with fishing boats and mostly within
306 EEZs with much lower boat densities and Crozet birds appeared to spend more time with fishing
307 boats both outside and within EEZs. Moreover, we have shown that large-scale overlap analyses
308 ($5 \times 5^\circ$) of AIS data was not related to the proxy of fine-scale bycatch risk (Table 3). It is
309 therefore very clear that the density of (fishing) boats within the foraging range does not linearly
310 translate into time spent attending boats.

311 This discrepancy between boat density in the foraging range of seabirds and the actual
312 time birds spent attending fishing boats calls for caution when estimating bycatch risk from
313 large-scale overlap data. AIS data is costly but it may be more easily accessible to researchers
314 than the often confidential and geographically-restricted VMS data (Votier et al., 2010) to allow
315 for fine-scale analyses. However large-scale overlap analyses will still be needed in particular
316 when bird tracking data is available at lower resolution than that offered by GPS tracking
317 devices (Clay et al., 2019a). This may be the case for many studies using GLS devices on non-
318 breeding individuals (juveniles, failed breeders, adults in winter or on sabbatical, etc.) or for
319 small species for which relatively large GPS device deployment could be problematic (Le Corre
320 et al., 2012; Delord et al., 2014). It would thus be useful to understand why a higher boat density
321 does not necessarily translate into more time spent by seabirds attending boats, and under what
322 circumstances this applies, to improve bycatch risk estimation from large-scale data.

323 An intriguing consideration raised by our results relevant to improving bycatch risk
324 estimations from large-scale data, is what we may call the dilution-shield hypothesis. An
325 increasing boat density may increase the encounter probability (Julien Collet, Patrick, &
326 Weimerskirch, 2017), but when a bird starts to attend and exploit a boat it reduces its
327 exploration time and chances of encountering further boats. Indeed, we clearly show that the
328 densest the boat seascape is, the lower the ratio of all boats in the seascape being actually
329 approached by birds (Table 1). Beyond a certain value, boat density may become less important

330 to consider than which boat(s) birds are attending. In the Mediterranean Sea, shearwaters were
331 observed to interact less with longliners when trawlers were present than when trawlers were
332 absent (Soriano-Redondo et al., 2016). In our study, Amsterdam albatrosses ranged by far in
333 the densest fishing boat seascape, but also in the densest non-fishing boat seascape. In contrast,
334 Kerguelen birds did not range in a very dense fishing boat area, but virtually all boats on the
335 Kerguelen shelf are fishing boats (Table 1). If birds are not strongly selective on the boats they
336 attend after encounters, non-fishing boats might dilute the bycatch risk and act as a shield
337 against more dangerous boats. Indeed we found that outside of EEZs, birds spent most of their
338 time in attendance of boats not associated with fisheries: 100% for Kerguelen, 93% for
339 Amsterdam, 81% for Marion and 35% for Crozet (within EEZs only fishing boats were
340 identified). Moreover, within all four populations, the ratio of fishing in relation to non-fishing
341 boats remained relatively similar across the three investigated scales (5, 30 and 100km, Table
342 1), suggesting low selectivity. It may be worth further investigating this hypothesis because if
343 this is correct, bycatch risk estimations from large scale overlap data (Clay et al., 2019; Heerah
344 et al., 2019) might relatively easily be improved by considering not only fishing boat density
345 but also non-fishing boat density within the foraging ranges of seabirds.

346 Boats may also cause other non-lethal issues beyond bycatch (nutritional and/or
347 dependence issues), especially if they become an important part of birds' time budgets (Fig. 2b)
348 and/or diet. To our knowledge, there are very few studies that have looked at how fisheries-
349 independent boats may impact foraging success and behavior in seabirds. However, it seems
350 that our studied birds spent a low fraction of their foraging time attending them. Of greater
351 concern is the large amount of time Kerguelen birds seem to spend attending the toothfish
352 longline fishery operating around their breeding ground. The nature, quality and amount of food
353 albatrosses can obtain from these toothfish fisheries are unclear considering bycatch mitigation

354 measures that are implemented. However, Kerguelen birds appear to be more dependent on
355 these boats than the most studied Crozet population.

356 An important question about bycatch risk is whether the large differences we observe in
357 the time incubating birds spent attending different types of fishing boats across populations may
358 be related to differences in population trends. Amsterdam albatrosses have been increasing
359 since the 1980s at a high rate, suggesting that they suffer minimal if any mortality from fisheries
360 (Rivalan, Barbraud, Inchausti, & Weimerskirch, 2010; Weimerskirch et al., 1997). Although
361 they forage in zones with high densities of both fishing and non-fishing boats, birds do not seem
362 to be particularly attracted by fishing boats: the low interaction to boats may explain why this
363 population has been able to increase steadily over the past four years. The three other
364 populations have shown similar trends until about 15 years ago, with a steep decline in the
365 1970s and early 1980 followed by a partial recovery (Nel et al., 2002; Weimerskirch et al.,
366 1997). Since then, the population on Marion has been increasing, whereas Kerguelen and Crozet
367 populations are stable (Ryan, Jones, Dyer, Upfold, & Crawford, 2009; Weimerskirch et al.,
368 2018). This difference in population dynamics of the wandering albatross populations could be
369 related to the lower encounter and attendance rates of Marion birds compared to Crozet and
370 Kerguelen birds.

371

372 Seabirds are one of the animal groups with the largest proportion of threatened species
373 and there has been much effort globally to better understand causative mechanisms behind
374 declining populations for conservation purposes. In this paper, we proposed a simple method
375 for estimating fine scale interactions between seabirds and boats with AIS. This method is
376 easily implemented through the combination of seabirds GPS tracks, now routinely collected

377 globally (Burger & Shaffer, 2008; Le Corre et al., 2012), and AIS data, which is readily
378 available (International Maritime Organisation).

379 With this method we provided the most direct and comprehensive assessment to date
380 of bycatch risk for large albatrosses breeding in the Indian Ocean, including for one of the
381 most threatened bird species. We illustrated the pros and cons of using AIS data for such
382 estimations, compared to other existing methods (large-scale overlap analyses and/or use of
383 embarked radar detectors). Importantly we showed that fishing boat density may not be a
384 good proxy to predict time spent attending boats and bycatch risk, and we proposed a general
385 hypothesis of shield effect from other types of boats to explain this discrepancy. Our results
386 also reveal extensive variations in the time and proportion of foraging time populations spent
387 attending various types of boats, which may cause other non-lethal issues beyond bycatch
388 risks, especially in the Kerguelen population.

389

390 **Acknowledgements**

391 The study is a contribution to the Program EARLYLIFE funded by a European Research
392 Council Advanced Grant under the European Community's Seven Framework
393 Program FP7/2007–2013 (Grant Agreement ERC-2012-ADG_20120314 to Henri
394 Weimerskirch) and to the Program Ocean Sentinel funded by the ERC under European
395 Community's H2020 Program (Grant Agreement ERC-2017-PoC_780058 to HW). The
396 fieldwork was also funded by program IPEV n° 109. South Africa's Department of
397 Environmental Affairs provided logistical support for fieldwork on Marion Island, which was
398 funded through the South African National Research Foundation (grant SNA93071 to Pierre
399 Pistorius).

400 Field procedures on Crozet, Kerguelen and Amsterdam were supported by the ‘Préfet of
401 Terres Australes et Antarctiques Françaises’. We thank the Reserve Nationale des TAAF for
402 funding loggers deployed on Amsterdam Island. We thank the fieldworkers involved in the
403 study on four sites, in particular Florent Lacoste, Tobie Getti, Aude Schreiber, Jérémy
404 Dechartre, Yusuke Goto, Yoshi Yonehara, Danielle and Stephan Keys. We thank Adrien
405 Pajot, for the help with AIS data. We thank the house ‘Ici c’est Olbreuse’ for providing
406 accommodation and fruitful discussions to half of the authors.

407 **Data accessibility**

408 Data are available in figshare at doi: 10.6084/m9.figshare.10289096 (Weimerskirch, Collet,
409 Corbeau, Pajot, Hoarau, Marteau, Filippi, Patrick, et al., 2019)

410 **Authors’ contributions**

411 HW conceived the project, AC, JC, FO, HW and PP contributed data and/or did field work
412 and prepared the data, JC merged AIS data to tracking data, AC performed all the other
413 analyses, AC and HW wrote the original paper and all authors commented on earlier drafts.

414

415 **References**

- 416 Anderson, O., Small, C., Croxall, J., Dunn, E., Sullivan, B., Yates, O., & Black, A. (2011).
417 Global seabird bycatch in longline fisheries. *Endanger. Species Res.* **14**, 91–106.
- 418 Belkin, I. M., & Gordon, A. L. (1996). Southern Ocean fronts from the Greenwich meridian
419 to Tasmania. *J. Geophys. Res. Oceans* **101**, 3675–3696.
- 420 Bertrand, S., Joo, R., Arbulu Smet, C., Tremblay, Y., Barbraud, C., & Weimerskirch, H.
421 (2012). Local depletion by a fishery can affect seabird foraging. (M. Frederiksen,
422 Ed.) *J. Appl. Ecol.* **49**, 1168–1177.

423 Bicknell, A. W. J., Oro, D., Camphuysen, K. C. J., & Votier, S. C. (2013a). Potential
424 consequences of discard reform for seabird communities. (J. Blanchard, Ed.)*J. Appl.*
425 *Ecol.* **50**, 649–658.

426 Bicknell, A. W. J., Oro, D., Camphuysen, K. C. J., & Votier, S. C. (2013b). Potential
427 consequences of discard reform for seabird communities. (J. Blanchard, Ed.)*J. Appl.*
428 *Ecol.* **50**, 649–658.

429 Brothers, N. (1991). Albatross mortality and associated bait loss in the Japanese longline
430 fishery in the Southern Ocean. *Biol. Conserv.* **55**, 255–268.

431 Brothers, N., Duckworth, A. R., Safina, C., & Gilman, E. L. (2010). Seabird Bycatch in
432 Pelagic Longline Fisheries Is Grossly Underestimated when Using Only Haul Data.
433 (S. J. Bograd, Ed.)*PLoS ONE* **5**, e12491.

434 Bugoni, L., McGill, R. A. R., & Furness, R. W. (2010). The importance of pelagic longline
435 fishery discards for a seabird community determined through stable isotope analysis.
436 *J. Exp. Mar. Biol. Ecol.* **391**, 190–200.

437 Burger, A. E., & Shaffer, S. A. (2008). APPLICATION OF TRACKING AND DATA-
438 LOGGING TECHNOLOGY IN RESEARCH AND CONSERVATION OF
439 SEABIRDS. *The Auk* **125**, 253–264.

440 Clark, B. L., Vigfúsdóttir, F., Jessopp, M. J., Burgos, J. M., Bodey, T. W., & Votier, S. C.
441 (2020). Gannets are not attracted to fishing vessels in Iceland—potential influence of a
442 discard ban and food availability. (K. Camphuysen, Ed.)*ICES J. Mar. Sci.* **77**, 692–
443 700.

444 Clay, T. A., Small, C., Tuck, G. N., Pardo, D., Carneiro, A. P. B., Wood, A. G., Croxall, J. P.,
445 Crossin, G. T., & Phillips, R. A. (2019). A comprehensive large-scale assessment of
446 fisheries bycatch risk to threatened seabird populations. (V. Paiva, Ed.)*J. Appl. Ecol.*

447 Collet, J, Patrick, S., & Weimerskirch, H. (2015). Albatrosses redirect flight towards vessels
448 at the limit of their visual range. *Mar. Ecol. Prog. Ser.* **526**, 199–205.

449 Collet, Julien, Patrick, S. C., & Weimerskirch, H. (2017). Behavioral responses to encounter
450 of fishing boats in wandering albatrosses. *Ecol. Evol.* **7**, 3335–3347.

451 Corbeau, A., Collet, J., Fontenille, M., & Weimerskirch, H. (2019). How do seabirds modify
452 their search behaviour when encountering fishing boats? (D. Hyrenbach, Ed.)*PLOS*
453 *ONE* **14**, e0222615.

454 Croxall, J. P., Butchart, S. H. M., Lascelles, B., Stattersfield, A. J., Sullivan, B., Symes, A., &
455 Taylor, P. (2012). Seabird conservation status, threats and priority actions: a global
456 assessment. *Bird Conserv. Int.* **22**, 1–34.

457 Cury, P. M., Boyd, I. L., Bonhommeau, S., Anker-Nilssen, T., Crawford, R. J. M., Furness, R.
458 W., Mills, J. A., Murphy, E. J., Osterblom, H., Paleczny, M., Piatt, J. F., Roux, J.-P.,
459 Shannon, L., & Sydeman, W. J. (2011). Global Seabird Response to Forage Fish
460 Depletion--One-Third for the Birds. *Science* **334**, 1703–1706.

461 Delord, K., Barbraud, C., Bost, C. A., Cherel, Y., Guinet, C., & Weimerskirch, H. (2014).
462 Atlas of top predators from French Southern Territories in the Southern Indian Ocean.
463 CNRS.

464 Delord, K., Gasco, N., Weimerskirch, H., Barbraud, C., & Micol, T. (2005). Seabird mortality
465 in the Patagonian toothfish longline fishery around Crozet and Kerguelen Islands,
466 2001–2003. *Ccamlr Sci.* **12**, 53–80.

467 Gremillet, D., Pichegru, L., Kuntz, G., Woakes, A. G., Wilkinson, S., Crawford, R. J. M., &
468 Ryan, P. G. (2008). A junk-food hypothesis for gannets feeding on fishery waste.
469 *Proc. R. Soc. B Biol. Sci.* **275**, 1149–1156.

470 Grémillet, D., Ponchon, A., Paleczny, M., Palomares, M.-L. D., Karpouzi, V., & Pauly, D.
471 (2018). Persisting Worldwide Seabird-Fishery Competition Despite Seabird
472 Community Decline. *Curr. Biol.* **28**, 4009-4013.e2.

473 Heerah, K., Dias, M. P., Delord, K., Oppel, S., Barbraud, C., Weimerskirch, H., & Bost, C. A.
474 (2019). Important areas and conservation sites for a community of globally threatened
475 marine predators of the Southern Indian Ocean. *Biol. Conserv.* **234**, 192–201.

476 Le Bot, T., Lescroël, A., & Grémillet, D. (2018a). A toolkit to study seabird–fishery
477 interactions. (S. Votier, Ed.) *ICES J. Mar. Sci.* **75**, 1513–1525.

478 Le Bot, T., Lescroël, A., & Grémillet, D. (2018b). A toolkit to study seabird–fishery
479 interactions. (S. Votier, Ed.) *ICES J. Mar. Sci.* **75**, 1513–1525.

480 Le Corre, M., Jaeger, A., Pinet, P., Kappes, M. A., Weimerskirch, H., Catry, T., Ramos, J. A.,
481 Russell, J. C., Shah, N., & Jaquemet, S. (2012). Tracking seabirds to identify potential
482 Marine Protected Areas in the tropical western Indian Ocean. *Biol. Conserv.* **156**, 83–
483 93.

484 Lewison, R., Crowder, L., Read, A., & Freeman, S. (2004). Understanding impacts of
485 fisheries bycatch on marine megafauna. *Trends Ecol. Evol.* **19**, 598–604.

486 Moore, J. K., Abbott, M. R., & Richman, J. G. (1999). Location and dynamics of the
487 Antarctic Polar Front from satellite sea surface temperature data. *J. Geophys. Res.*
488 *Oceans* **104**, 3059–3073.

489 Nel, Deon. C., Ryan, P. G., Nel, J. L., Klages, N. T. W., Wilson, R. P., Robertson, G., &
490 Tuck, G. N. (2002). Foraging interactions between Wandering Albatrosses *Diomedea*
491 *exulans* breeding on Marion Island and long-line fisheries in the southern Indian
492 Ocean: Foraging interactions in albatrosses. *Ibis* **144**, E141–E154.

493 Oro, D., Genovart, M., Tavecchia, G., Fowler, M. S., & Martínez-Abraín, A. (2013).
494 Ecological and evolutionary implications of food subsidies from humans. (B. Worm,
495 Ed.)*Ecol. Lett.* **16**, 1501–1514.

496 Pante, E., & Simon-Bouhet, B. (2013). marmap: A Package for Importing, Plotting and
497 Analyzing Bathymetric and Topographic Data in R. (G. J.-P. Schumann, Ed.)*PLoS*
498 *ONE* **8**, e73051.

499 Pauly, D., Christensen, V., Guénette, S., Pitcher, T. J., Sumaila, U. R., Walters, C. J., Watson,
500 R., & Zeller, D. (2002). Towards sustainability in world fisheries. *Nature* **418**, 689–
501 695.

502 Phillips, R. A., Xavier, J. C., & Croxall, J. P. (2003). Effects of satellite transmitters on
503 albatrosses and petrels. *The Auk* **120**, 1082–1090.

504 Pirotta, E., Edwards, E. W. J., New, L., & Thompson, P. M. (2018). Central place foragers
505 and moving stimuli: A hidden-state model to discriminate the processes affecting
506 movement. (L. Börger, Ed.)*J. Anim. Ecol.* **87**, 1116–1125.

507 Read, A. J. (2008). The looming crisis: interactions between marine mammals and fisheries. *J.*
508 *Mammal.* **89**, 541–548.

509 Rivalan, P., Barbraud, C., Inchausti, P., & Weimerskirch, H. (2010). Combined impacts of
510 longline fisheries and climate on the persistence of the Amsterdam Albatross
511 *Diomedea amsterdamensis*. *Ibis* **152**, 6–18.

512 Ryan, P. G., Jones, M. G., Dyer, B. M., Upfold, L., & Crawford, R. J. (2009). Recent
513 population estimates and trends in numbers of albatrosses and giant petrels breeding at
514 the sub-Antarctic Prince Edward Islands. *Afr. J. Mar. Sci.* **31**, 409–417.

515 Soriano-Redondo, A., Cortés, V., Reyes-González, J. M., Guallar, S., Bécáres, J., Rodríguez,
516 B., Arcos, J. M., & González-Solís, J. (2016). Relative abundance and distribution of
517 fisheries influence risk of seabird bycatch. *Sci. Rep.* **6**, 37373.

518 Thiebault, A., Mullers, R. H. E., Pistorius, P. A., & Tremblay, Y. (2014). Local enhancement
519 in a seabird: reaction distances and foraging consequence of predator aggregations.
520 *Behav. Ecol.* **25**, 1302–1310.

521 Thiebot, J.-B., Delord, K., Barbraud, C., Marteau, C., & Weimerskirch, H. (2015). 167
522 individuals versus millions of hooks: bycatch mitigation in longline fisheries underlies
523 conservation of Amsterdam albatrosses: Amsterdam Albatrosses and Longline
524 Fisheries. *Aquat. Conserv. Mar. Freshw. Ecosyst.* n/a-n/a.

525 Torres, L., Sagar, P., Thompson, D., & Phillips, R. (2013). Scaling down the analysis of
526 seabird-fishery interactions. *Mar. Ecol. Prog. Ser.* **473**, 275–289.

527 Votier, S. C., Bearhop, S., Witt, M. J., Inger, R., Thompson, D., & Newton, J. (2010).
528 Individual responses of seabirds to commercial fisheries revealed using GPS tracking,
529 stable isotopes and vessel monitoring systems. *J. Appl. Ecol.* **47**, 487–497.

530 Votier, S. C., Furness, R. W., Bearhop, S., Crane, J. E., Caldow, R. W. G., Catry, P., Ensor,
531 K., Hamer, K. C., Hudson, A. V., Kalmbach, E., Klomp, N. I., Pfeiffer, S., Phillips, R.
532 A., Prieto, I., & Thompson, D. R. (2004). Changes in fisheries discard rates and
533 seabird communities. *Nature* **427**, 727–730.

534 Weimerskirch, H., Collet, J., Corbeau, A., Pajot, A., Hoarau, F., Marteau, C., Filippi, D., &
535 Patrick, S. C. (2020). Ocean sentinel albatrosses locate illegal vessels and provide the
536 first estimate of the extent of nondeclared fishing. *Proc. Natl. Acad. Sci.* 201915499.

537 Weimerskirch, H., Collet, J., Corbeau, A., Pajot, A., Hoarau, F., Marteau, C., Filippi, D.,
538 Patrick, S. C., Pistorius, P., Orgeret, F., & Joo, R. (2019).
539 Data_OCEAN_SENTINEL_Weimerskirch_et_al_2018_2019. *Figshare - Dataset*.

540 Weimerskirch, H., Filippi, D. P., Collet, J., Waugh, S. M., & Patrick, S. C. (2017). Use of
541 radar detectors to track attendance of albatrosses at fishing vessels: Seabird-Fishery
542 Interactions. *Conserv. Biol.*

543 Weimerskirch, Henri, Brothers, N., & Jouventin, P. (1997). Population dynamics of
544 wandering albatross *Diomedea exulans* and Amsterdam albatross *D. amsterdamensis*
545 in the Indian Ocean and their relationships with long-line fisheries: conservation
546 implications. *Biol. Conserv.* **79**, 257–270.

547 Weimerskirch, Henri, Delord, K., Barbraud, C., Le Bouard, F., Ryan, P. G., Fretwell, P., &
548 Marteau, C. (2018). Status and trends of albatrosses in the French Southern
549 Territories, Western Indian Ocean. *Polar Biol.* **41**, 1963–1972.

550

Table

Table 1 – Differences between each site: mean and standard deviation of parameters and significance letters of Tuckey tests (same letter in different site mean no difference and different letters mean significant differences).

	Amsterdam (trips: n=10, 10 complete)			Crozet (trips: n=49, 48 complete)			Kerguelen (trips: n=27, 23 complete)			Marion (trips: n=57, 53 complete)		
	Mean	Standard deviation	Significance letter	Mean	Standard deviation	Significance letter	Mean	Standard deviation	Significance letter	Mean	Standard deviation	Significance letter
PER TRIP (n=143)												
Maximum distance from the colony (km)	1637.70	1281.10	AB	1177.90	813.90	AB	886.60	711.85	A	1424.70	660.54	B
Trip duration (day)	9.53	3.48	A	11.09	5.60	A	10.60	4.14	A	12.60	4.46	A
Ratio in EEZ	0.39	0.41	AC	0.57	0.35	AB	0.74	0.32	B	0.36	0.24	C
Ratio on shelf	0.20	0.29	AC	0.40	0.33	A	0.65	0.29	B	0.11	0.15	C
Ration in Antarctic waters	0.00	0.00	A	0.07	0.20	A	0.19	0.30	A	0.12	0.26	A
Ration in subantarctic waters	0.03	0.07	A	0.72	0.34	B	0.78	0.30	B	0.57	0.31	B
Ration in subtropical waters	0.97	0.07	A	0.21	0.32	BC	0.02	0.13	B	0.31	0.32	C
Number of encounters	5.10	7.61	A	3.06	2.18	A	2.89	2.65	A	4.23	5.45	A
Time in encounter (h)	19.04	23.28	A	17.08	18.36	A	53.30	62.82	B	11.91	21.96	A
Number of attendances	1.80	2.20	A	1.71	1.57	A	3.59	4.41	A	1.72	2.70	A
Time in attendance (h)	4.31	6.97	A	6.75	11.00	A	31.14	38.22	B	3.21	10.20	A
Time in attendance with fishing vessels (h)	3.16	7.56	A	6.68	11.31	A	30.63	39.27	B	2.64	10.31	A
Time in attendance in EEZ (h)	2.90	6.71	A	5.84	11.14	A	28.37	38.60	B	2.46	10.24	A
Time in attendance out EEZ (h)	1.42	2.40	A	0.91	2.74	A	2.77	11.56	A	0.75	1.58	A
Time in attendance with fishing vessels in EEZ (h)	2.82	6.75	A	5.84	11.14	A	28.36	38.60	B	2.45	10.24	A
Time in attendance with fishing vessels out EEZ (h)	0.11	0.28	A	0.60	2.66	A	0.00	0.00	A	0.14	0.68	A
Ratio of fishing vessels attended (5km)	0.64	0.43	AB	0.76	0.42	AB	0.95	0.23	A	0.22	0.37	B
Ratio of fishing vessels encountered (30km)	0.63	0.36	AB	0.76	0.40	A	0.92	0.28	A	0.17	0.32	B
Ratio of fishing vessels in seascape (100km)	0.43	0.27	A	0.75	0.36	B	0.92	0.23	B	0.16	0.29	A
PER LOCATION (n=619631)												
Number of boats attended (5km)	0.02	0.16	A	0.02	0.15	A	0.11	0.31	B	0.01	0.11	C
Number of boats encountered (30km)	0.19	0.84	A	0.06	0.26	B	0.23	0.43	C	0.05	0.26	B
Number of boats in seascape (100km)	1.99	5.08	A	0.29	0.60	B	0.40	0.60	C	0.35	1.11	D
Number of fishing vessels in seascape (100km)	0.83	1.93	A	0.21	0.51	B	0.39	0.60	C	0.07	0.33	D
Ratio of number of boats 5 km / 100 km	0.04	0.19	A	0.07	0.24	A	0.30	0.45	B	0.04	0.18	A

55 **Table 2** – Number (and percentage) of trips per site with boats in seascapes (100km), encountered (30km) and attended (5km).

	Amsterdam (n=10)	Crozet (n=49)	Kerguelen (n=27)	Marion (n=57)	TOTAL (n=143)
With boats in seascape (100km)	9 (90%)	48 (97.96%)	27 (100%)	39 (68.42%)	123 (86.01%)
With boats encountered (30km)	8 (80%)	42 (85.71%)	23 (85.19%)	36 (63.16%)	109 (76.22%)
With boats attended (5km)	6 (60%)	36 (73.47%)	18 (66.67%)	27 (47.37%)	87 (60.84%)

56

57

58 **Table 3** – Correlations (with p value of Pearson test) between the number of AIS signals (total and for fishing boats only) per 5x5° grid cells and the time spent
59 by birds in same 5x5° grid cell (for a total time, for time with boats in seascape (<100km), for time with boats in encounter (<30km) and for time with boats in
60 attendance (<5km)) for the active months of the different populations of albatrosses.

<u>ANALYSES FOR 5x5° GRID</u>	Amsterdam (n=37 cells)		Crozet (n=63 cells)		Kerguelen (n=41 cells)		Marion (n=54 cells)	
	Number of AIS signals (April)	Number of AIS fishery signals (April)	Number of AIS signal (January & February)	Number of AIS fishery signals (January & February)	Number of AIS signals (January & February)	Number of AIS fishery signals (January & February)	Number of AIS signals (February & March)	Number of AIS fishery signals (February & March)
Bird time spent in grid cell	-0.113 (p=0.52)	-0.113 (p=0.52)	-0.094 (p=0.48)	-0.094 (p=0.48)	-0.039 (p=0.81)	-0.056 (p=0.73)	-0.012 (p=0.94)	-0.012 (p=0.94)
Bird time spent with boat at 100km	0.062 (p=0.72)	0.062 (p=0.73)	-0.021 (p=0.87)	-0.021 (p=0.88)	0.026 (p=0.87)	0.008 (p=0.96)	0.199 (p=0.18)	0.199 (p=0.19)
Bird time spent in encounter	0.194 (p=0.27)	0.192 (p=0.28)	-0.029 (p=0.83)	-0.029 (p=0.83)	0.025 (p=0.88)	0.007 (p=0.97)	0.164 (p=0.28)	0.165 (p=0.28)
Bird time spent in attendance	-0.050 (p=0.78)	-0.052 (p=0.77)	-0.038 (p=0.78)	-0.037 (p=0.78)	0.018 (p=0.91)	0.002 (p=0.99)	-0.014 (p=0.93)	-0.014 (p=0.93)

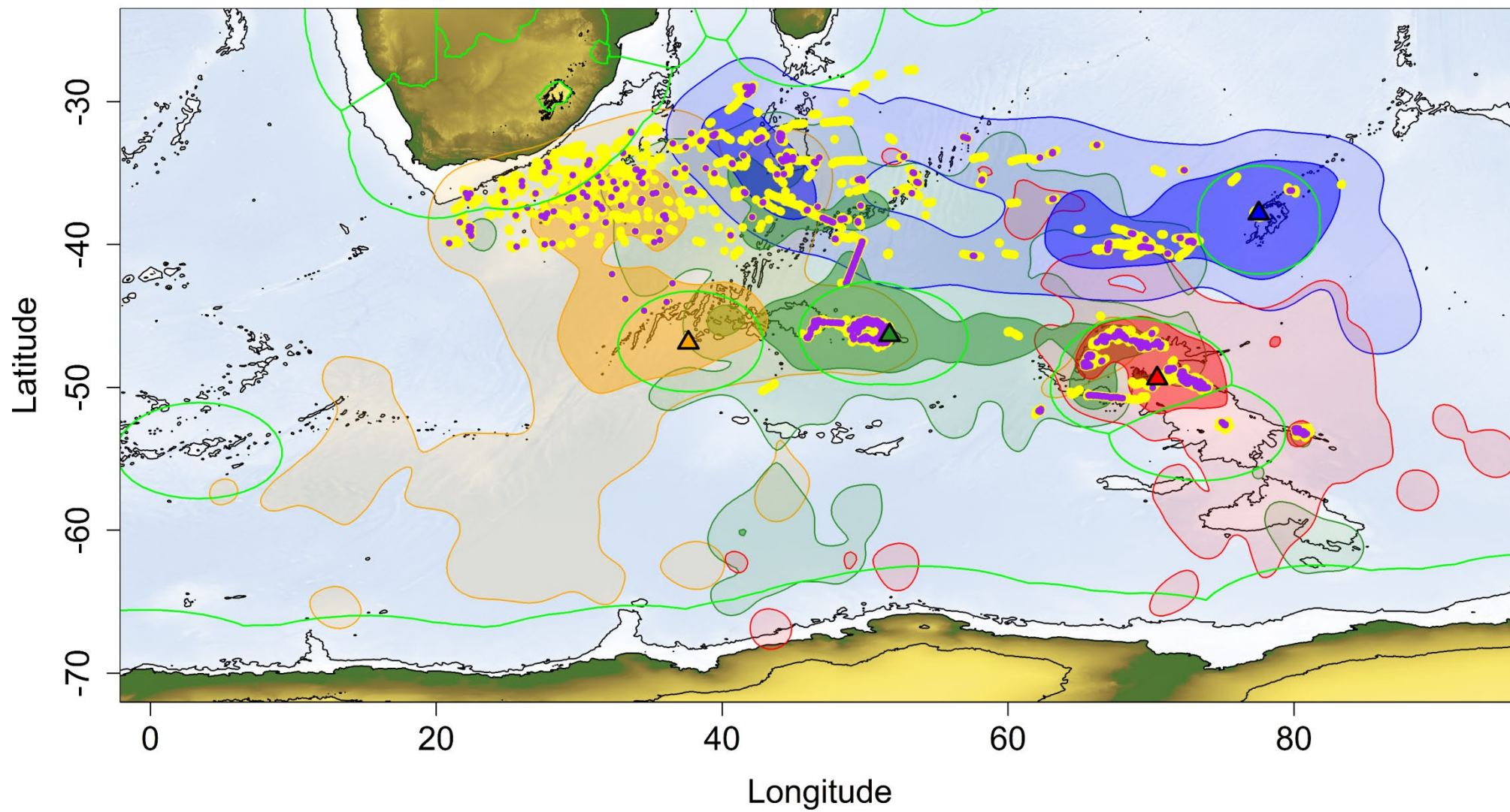
61

62 **Figures legends**

63 **Figure 1** - Map of the South Indian Ocean with kernel utilization distribution 50% (darker shade) and 90%
64 (lighter shade) of birds for each site (triangles) (blue = Amsterdam, green = Crozet, red= Kerguelen, orange=
65 Marion); yellow dots represent encounter events and purple dots, attendance events; isobaths: -2000 m (shelf),
66 0 m and +2000 m; light-green lines represent EEZ.

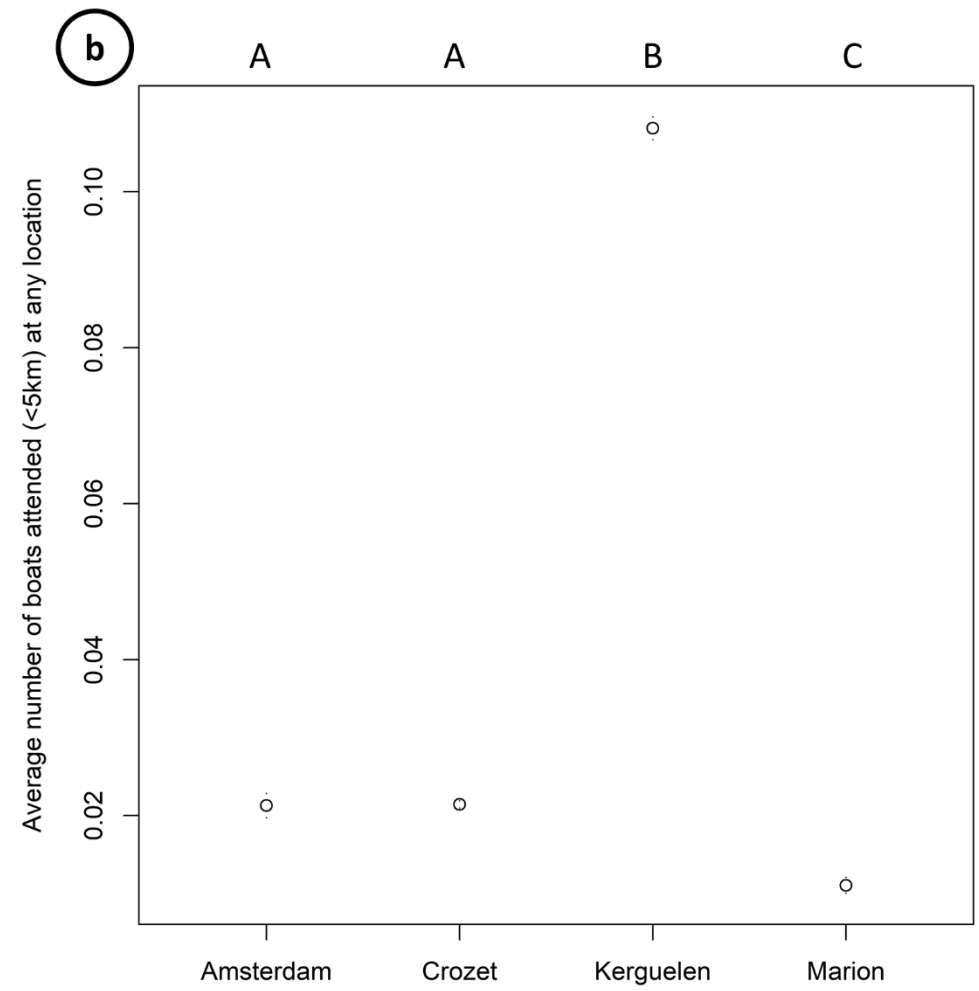
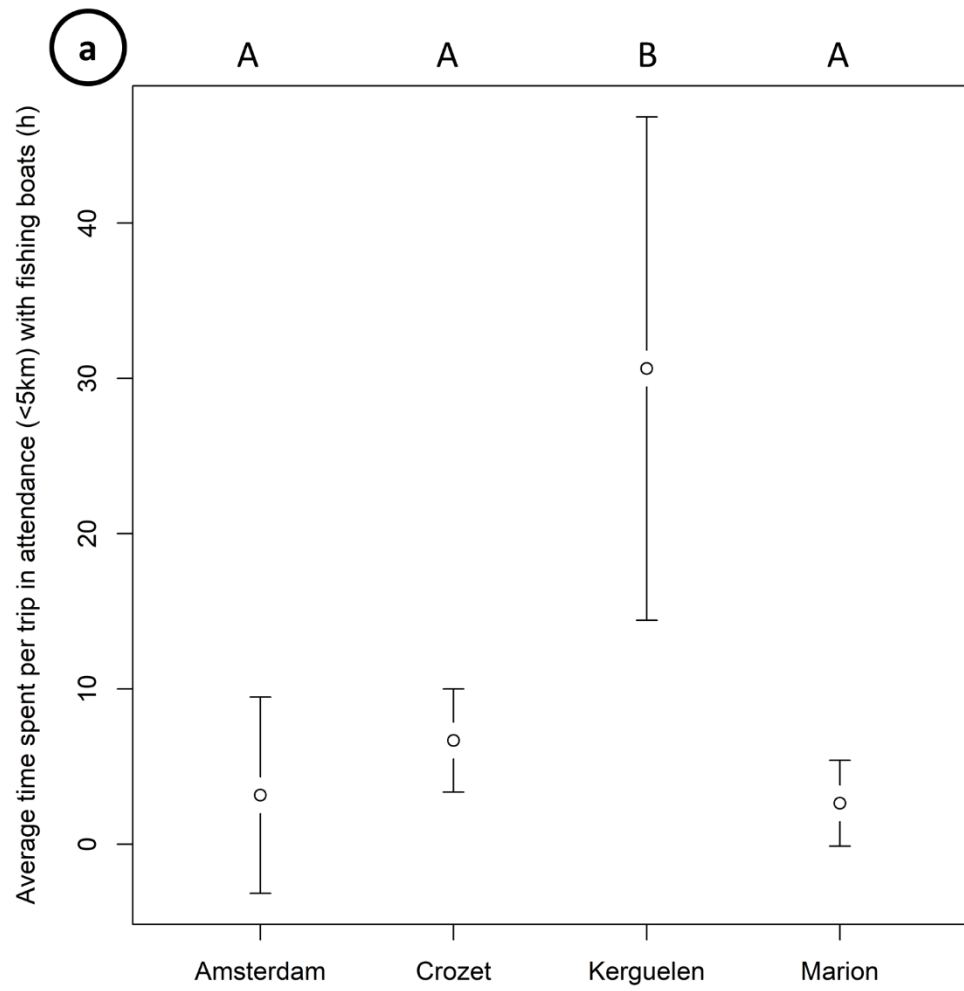
67 **Figure 2** – Proxy of bycatch risk as **(a)** time spent per trip in attendance (within 5km) with fishing boats
68 (hours) and **(b)** Number of boat attended (within 5 km) at any location; Mean and confidence interval (95%)
69 of each site. Letters represent significant difference.

70



572

573 **Figure 1**



574

575 **Figure 2**