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Fisheries bycatch mitigation measures as an efficient tool for the conservation of seabird populations

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Abstract

10

11   1. The impact of industrial fisheries on marine biodiversity is conspicuous in large pelagic vertebrate's

12   fisheries bycatch. In seabirds, this led to the decline of many populations since the 1980's following the rise

13   of global fishing effort. Bycatch mitigation measures were implemented since the 2000's, but their effects

14   on the concerned seabird populations remain poorly quantified and understood.

15   2. We studied the effects of bycatch mitigation measures on the demography of the white-chinned petrel,

16   one of the most bycatch impacted seabirds whose populations suffered dramatic declines before the

17   implementation of mitigation measures. To do so we 1) built multi-event capture-recapture models to

18   estimate the demographic parameters of a population from Possession Island (southern Indian Ocean) over

19   thirty years, 2) assessed the effect of climate and fishery covariates on demographic parameters, 3) built a

20   population matrix model to estimate stochastic growth rate according to the management in fisheries

21   bycatch, and 4) estimated changes in breeding population density using distance sampling data.

22   3. The population declined from the 1980's to the mid-2000's, while trawl and longline fisheries occurred

23   with no bycatch mitigation measures. The negative effects of fishery bycatch through additive mortality and

24   of rat predation on breeding success were likely the main drivers of this decline.

25   4. Both modelled population growth rate and observed breeding densities showed an increase since the mid-

26   2000's. We explained this trend by the improvement in survival probability following implementation of

27   fishery bycatch mitigation measures and in breeding success probability with the local control of the rat

28   population and changes in sea ice conditions on foraging grounds.

29   6. Synthesis and applications: We provide a holistic approach to assess the effects of management measures

30   by analysing datasets from sampling methods commonly employed in seabirds studies. Our conclusions

31   should encourage the eradication of invasive predatory species in seabirds breeding areas and the

32   strengthening of bycatch mitigation measures for the vulnerable seabird species, especially in international

33   waters, but also the development of such measures considering the other marine large pelagic species

34   threatened by fisheries bycatch (sharks, rays, turtles and marine mammals) since it could be crucial to avoid

35   populations' extinction.

36   **Keywords**

Introduction

Managing industrial fishing activities remains an urgent and crucial challenge for marine biodiversity conservation and food security (Zhou et al., 2010). Massive and selected marine resources exploitation causes a considerable impact on marine fish stocks and ecosystems' structures (Pauly et al., 2005), causing death by accidentally catching non-targeted species of pelagic vertebrate top-predators (sharks, turtles, marine mammals and seabirds) whose populations have severely declined in the last decades (Lewison et al., 2014; Nel et al., 2002; Stevens et al., 2000). Large marine vertebrates play a major role on the stability of trophic webs (Estes et al., 2011) and provide unique evolutionary innovations (Kelley & Pyenson, 2015). To reduce this impact, bycatch mitigation measures were developed, tested and implemented since the early 2000's. They have led to substantial reductions in bycatch of large marine vertebrates in several fisheries worldwide (Cox et al., 2007; Phillips et al., 2016). However, the long-term efficiency of bycatch mitigation measures on trends and abundances of concerned populations may be challenging to estimate and has been poorly assessed.

Seabird conservation state is of concern with 28% of their species globally threatened (Croxall et al., 2012) and 66% of albatrosses and large petrels species actually listed by the Agreement on the Conservation of Albatrosses and Petrels (ACAP) as a priority (Phillips et al., 2016). Fishery bycatch has been identified as the most impacting threat on those species and alien species predation as the threat that affects the more numerous of those species (Dias et al., 2019). Many of them share their foraging areas with fishing vessels that they follow – because Procellariforms use their sense of smell to track food – in order to take advantage of food opportunities such as fishing waste and baits. This increases hooking or entanglement risks and leads to the death of hundreds of thousands of seabirds every year (Tuck et al., 2003; Žydelis et al., 2013). This massive additive mortality is highly problematic for long-lived marine pelagic species such as seabirds characterised by slow demographic strategies due to long generation time, low fecundity and delayed maturity (Hamer et al., 2001). Their populations are sensitive to small variations in adult mortality (Caswell, 2001). Since the early 2000's, the scientific community, the non-governmental organisations, the countries

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64 that have ratified the ACAP and the Regional Fisheries Management Organisations (RFMOs) spent

65 considerable efforts to develop and implement effective mitigation measures in order to significantly reduce

66 or completely remove this threat. Measures include bird-scaring streamer lines, line weighting, night setting,

67 and establishment of protected areas (Løkkeborg, 2011; Wolfaardt et al., 2016). Unfortunately, many

68 threatened species do not yet beneficiate sufficient mitigation measures implementation (Gilman et al.,

69 2014).

70 Many studies showed the potential or direct efficacy of bycatch mitigation measures (Gilman, 2011; Maree

71 et al., 2014), but little is known about their long-term effect on population dynamics and trends (Forney et

72 al., 2020; van Beest et al., 2017). Furthermore, none of the studies investigated the retrospective efficiency

73 of these measures on seabird demography and population dynamics. This knowledge is yet crucial to

74 enhance the measures' implementation and to allow management optimisation of threatened populations

75 (van Beest et al., 2017). Assessing the retrospective effect of bycatch mitigation measures could be

76 challenging because of the complexity of the targeted species' phenology and their large scale at-sea

77 distributions, the need of long-term monitoring data necessary to detect demographic changes, and the need

78 to account for other factors that can potentially affect their population dynamics. Indeed, seabird populations

79 can also be threatened by climate change, human disturbance, pollution, diseases and alien introduced

80 predators (Croxall et al., 2012). Moreover, as long as fishing effort data remain sparse and inaccurate due

81 to insufficient global fisheries control, especially in international waters, highlighting the effect of fishing

82 activities on seabird populations remain difficult (Genovart et al., 2017) as well as the implementation of

83 bycatch mitigation measures.

84 Our aim was to evaluate the impact of fishery bycatch and implementation of mitigation measures on a

85 vulnerable seabird, the white-chinned petrel (*Procellaria aequinoctialis*), which is one of the most

86 accidentally captured petrel in pelagic fisheries targeting tuna species in South Atlantic and Indian Oceans,

87 and in demersal longline fisheries practices targeting Patagonian toothfish (*Dissostichus eleginoides*) in

88 South Indian Ocean and in Southern Ocean (Weimerskirch et al., 1999). The species was also largely

89   threatened by trawl fisheries that occurred in their sub-tropical wintering areas (Croxall, 2008) and in

90   subantarctic waters until the mid-1990's (Watkins et al., 2008; Weimerskirch et al., 2000).

91   We took advantage of long-term capture-recapture and population datasets of a white-chinned petrel

92   population breeding at Possession Island, Crozet archipelago, to i) estimate demographic parameters before

93   and after the implementation of bycatch mitigation measures in the subantarctic French exclusive economic

94   zone (EEZ) and in waters managed by the Commission for the Conservation of Antarctic Marine Living

95   Resources (CCAMLR); ii) test for temporal trends in demographic parameters and assess the relationship

96   with fishing efforts and climate variables; iii) model the population dynamics and assess the effect of the

97   implementation of bycatch mitigation measures and of the control of an introduced predator on the

98   population growth rate; and iv) estimate breeding population density using distance sampling data from

99   line-transect surveys conducted on land in 2011 and 2017.

100   There is evidence that longline fisheries bycatch, climate variability and introduced predators (black rat

101   *Rattus rattus*) impact demographic parameters and population growth rate of the white-chinned petrels from

102   the Crozet archipelago (Barbraud et al., 2008; Jouventin et al., 2003). The acceptation of fishery bycatch as

103   a serious threat by the CCAMLR emerged in the early 1990's, followed by the implementation of strong

104   bycatch mitigation measures since 2003 (Delord et al., 2010; Reid et al., 2010; Delord et al., 2005). In the

105   meantime, the ACAP was ratified by 13 countries including France that integrated it in French law in 2005.

106   Under the hypothesis that bycatch was the main cause of the white-chinned petrel population's decline

107   through additive mortality (Barbraud et al., 2008), and since numbers of white-chinned petrels caught in

108   longlines strongly decreased following the implementation of mitigation measures (CCAMLR, 2020;

109   Delord et al., 2010), we expected i) an increase in survival probability due to reduced direct bycatch

110   mortality, ii) an increase in breeding success since a widowed parent due to its partner death in fisheries

111   may fail to fledge the chick on its own, and iii) a stability or increase in the white-chinned petrel breeding

112   population density.

113

Material and Methods

Study species and population monitoring

The Ethics Committee of IPEV and the Comité Environnement Polaire approved the field procedures for the French Southern Territories. White-chinned petrels were monitored at Possession island (46° S ; 51° E) (see Appendix S1 in Supporting Information) following two sampling approaches. First, 200 individually marked burrows were investigated each year from the breeding season 1985/1986 (hereafter 1985) to the breeding season 2017/2018 at the study colony of Station de Pompage. Each burrow was visited several times from egg laying (November) to chick fledging (April) to assess the presence of the occupying individuals, their identity, and their breeding status (breeder if an egg was laid, non-breeder otherwise). Each adult newly captured and all fledglings were marked with a stainless-steel ring on the leg (Barbraud et al., 2008, see Appendix S2). Black rats were introduced at Possession Island during the 19th century (Chapuis et al., 1994) and were poisoned yearly during each breeding season since 1994 at the study colony but not elsewhere on Possession Island (Jouventin et al., 2003). Second, in order to estimate breeding population density, line-transect distance sampling surveys were carried out in 2011 and 2017 at Pointe Basse, that hosts a large proportion (25-30%) of white-chinned petrels breeding on Possession Island with relatively high densities of burrows (Barbraud et al., 2008). Surveys were conducted during incubation (early December) and densities of occupied burrows were estimated from line transect distance sampling (see Appendix S3).

Modelling demographic parameters

For individuals marked as adults, their encounter history started at first capture. The 32-year time series (from 1986 to 2017) resulted in 842 individual capture histories. To model the transition of individuals in state i at time t to state j at time t+1, we estimated the following demographic parameters (Figure 1, Table 1 S4): apparent survival probability (s), the state-specific transitional probability of becoming or remaining a breeder (hereafter breeding probability) (β), success probability (γ), recapture probability (p), and

139   assignment probability (δ). To do so, we constructed a multi-event capture-mark-recapture (MECMR)

140   model (Pradel, 2005) to take account of uncertainty in breeding success, with four states consisting of one

141   non-breeder state among adults (excluding subadults) (NB), two breeder states [failed breeder (FB),

142   successful breeder (SB)], and the dead state (†) that was not observable. Capture histories were coded

143   considering field observations: 0 = not observed, 1 = seen as NB, 2 = seen as FB, 3 = seen as SB, 4 = seen

144   as breeder but status not ascertained. We started with a general model as the model that allowed the most

145   numerous demographic parameters to be estimated independently for each step of time and for each state

146   (see Appendix S4).

147   For individuals marked as chick, their encounter history started at the first return in the study population,

148   from which they were considered as adults. Youngest individuals seen as breeders on the colony were five

149   years old. To estimate juvenile survival and recruitment probabilities we used data from individuals that

150   were marked as chicks only. Our dataset contained 979 individual capture histories of which 61 were seen

151   recruiting. We estimated the following parameters: apparent survival probability (s_n), the probability of first

152   breeding (r_n), as the probability that an individual breeds for the first time (Figure 1), and the recapture

153   probability. To do so, we constructed a multi-state capture-mark-recapture (MSCMR) model (Lebreton et

154   al., 2009; Pradel, 1996) with three states consisting of one juvenile state, one breeder state and the dead state

155   (see Appendix S4). The goodness of fit tests (GOF) for MSCMR and MECMR models (Pradel et al., 2003)

156   were performed using U-CARE 2.3.4 (Choquet et al., 2009) (see Appendix S4).

157

158   <u>Modelling population dynamics</u>

159   To model the dynamics of the white-chinned petrel population and the effects of different scenarios on the

160   population growth rate (λ), we built a female-only pre-breeding census matrix population model structured

161   by age and breeding status (Caswell, 2001) with seven juvenile age classes and three adult classes according

162   to breeding states (NB, SB and FB, see Appendix S5). From this matrix model we performed a sensitivity

163   analysis of λ to changes in demographic parameters, and a retrospective analysis to quantify the contribution

164   of the vital rates to variability in λ (Caswell, 2001).

165   Then, we estimated λ using six management scenarios according to bycatch mitigation measure

166   implementation and rat control (see Appendix S5, Table 1 S5). For each scenario the demographic

167   parameters s, β and γ differed following three periods: 1986-1993 (eight years), when trawl fishing occurred

168   and no rat control occurred; 1994-2004 (11 years), when longline fishing occurred without mitigation

169   measures and rat control occurred; 2005-2017 (13 years), when longline fishing occurred with mitigation

170   measures and rat control occurred. All other parameters remained equal. The first three scenarios had no

171   mitigation measures for fishery bycatch. Scenario (1) used s, β and γ estimated from 1986 to 1993. Scenario

172   (2) used s and β estimated from 1994 to 2004 and γ estimated from 1986 to 1993. Scenario (3) used s, β and

173   γ estimated from 1994 to 2004. We then built three additional scenarios with mitigation measures for

174   fisheries bycatch. We chose 2005 as the year-reference for the implementation of bycatch mitigation

175   measures because France integrated the ACAP recommendations in the French law in 2005. Scenario (4)

176   used s and β estimated from 2005 to 2017 and γ estimated from 1984 to 1993 (no rat control). Scenario (5)

177   used s, β and γ estimated from 2005 to 2017. Scenario (6) used β and γ estimated from 2005 to 2017, and s

178   for FB and SB individuals estimated as the intercept in the linear relationship between survival and demersal

179   longline fishing effort. This last scenario reflected population dynamics without any additive effect of

180   demersal longline fishing on adult survival.

181   Environmental stochasticity was modelled using the mean and process variance values of the demographic

182   parameters. We calculated them from the values of demographic parameters estimated by MECMR model

183   and following Gould & Nichols (1998), but ignoring the sampling covariance terms in the calculation of

184   temporal variation in the demographic parameters.

185   We conducted random effect design Life Table Response Experiments (LTRE) in a retrospective analyses

186   in order to estimate the contribution of each demographic parameter to the variability in the growth rate λ.

187   Following Jenouvrier et al., (2005), we approximated the contributions using the expression $V(\lambda) \approx s(\theta)^2$

188    $var(\theta)$, where $s(\theta)$ is the sensitivity of λ to the demographic parameter.

189

190

191   <u>Effects of covariates on demographic parameters</u>

192   We chose fisheries and climate covariates that were likely to have an influence on survival, reproduction or

193   breeding parameters, according to white-chinned petrel breeding phenology, distribution at sea (see

194   Appendix S6, Table 1 S6, Figures 1 S6, 2S6, 3 S6 & 4 S6) and previous knowledge of the studied population

195   (Barbraud et al., 2008; Catard et al., 2000; Delord et al., 2013; Péron et al., 2010). Fishery covariates

196   included fishing effort from demersal longline during the breeding season, pelagic longline during wintering

197   (i.e. the non-breeding season), illegal, unreported and unregulated (IUU) from Crozet EEZ (CCAMLR,

198   2018), and the quantity of hakes (*Merluccius spp.*) fished in Namibian EEZ used as a proxy of trawling

199   effort. Climate data included sea surface temperature anomalies (SSTa) and sea surface temperature (SST)

200   during the breeding and non-breeding seasons, sea ice concentration (SIC) during the breeding season, ,

201   chlorophyll a concentration (Chla) during the non-breeding season, southern annular mode (SAM) and

202   southern oscillation index (SOI) without lag and with a lag of one year (see Appendix S6, Figure 2 S6).

203   Relationships between covariates and demographic parameters were fitted using a logit link function:

204    $logit(\theta) = B0 + B1*Xn$ where θ is the demographic trait, B0 is an intercept parameter, B1 is a slope

205   parameter and Xn is the value of the covariate at year n (see Appendix S7). As demographic parameters

206   may not respond linearly to a certain climate driver, we also tested for quadratic effects of climate covariates.

207   When for a given demographic parameter a significant effect of a fishery and of a climate covariate was

208   detected, we also tested for the effect of an interaction between the fishery and the climate covariate.

209   Environmental variables were standardised by subtracting the mean and dividing by the standard error, in

210   order to help with model fitting. We performed an environmental sensitivity analysis to assess to which

211   covariate λ was the most sensitive (see Appendix S9).

212

Results

<u>Breeding population density</u>

Burrow occupancy rate was 0.889 ± 0.022 (SE) in 2011 and 0.921 ± 0.017 in 2017. Breeding population density at Pointe Basse was 6.54 active burrows/ha in 1983 and 3.09 active burrows/ha in 2004 (Marteau, 2008). We found that breeding population density at Pointe Basse was 5.65 ± 0.89 (N = 119) in 2011 and 5.68 ± 1.07 active burrows/ha (N = 126) in 2017 (see Appendix S3, Table 1 S3).

219

<u>Demographic parameters</u>

The GOF tests of the MECMR model did not indicate a lack of fit ($\chi^2 = 432.700$, df = 417, p = 0.288). The GOF tests of the MSCMR indicated a lack of fit ($\chi^2 = 603.588$, df = 82, p < 0.001) revealing a trap-dependence, so we used a coefficient of overdispersion ($\hat{c} = 2.58$) for model selection.

The mean observed age at first breeding was 7.3 ± 1.5 years old (median = 7), 82% of the first-time breeders were eight years old or less, 60.7% were seven years old or less and 34.4% were six years old or less. MSCMR model selection suggested that recruitment started at age five and was 0.061 ± 0.030, increased at 0.314 ± 0.082 at six years old and was 0.370 ± 0.130 at seven years old and older (Table 2 S4). The selected model suggested constant survival probability among age classes from seven years old at 0.879 ± 0.022. Mean annual survival probability was estimated from fledging to five years old at 0.781 ± 0.028.

MECMR model selection indicated that recapture probability was time and state dependent, and was 0.785 ± 0.250 for successful breeders, 0.863 ± 0.180 for failed breeders and 0.047 ± 0.006 for non-breeders. Assignment probability was 0.971 ± 0.004 for failed breeders and was ≈ 1 for successful breeders. Survival probability was time and state dependent (Figure 2). Mean survival probability of non-breeders was 0.904 ± 0.017, 0.870 ± 0.099 for successful breeders and 0.844 ± 0.106 for failed breeders. No linear trend was detected in survival probabilities (Table 3 S4).

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236 Breeding probability was time and state dependent and was higher for successful and failed breeders (0.870

237 ± 0.149 and 0.825 ± 0.147 respectively) than for non-breeders (0.350 ± 0.253) (Figure 2). A positive linear

238 trend was detected for failed breeders (slope = 0.391 ± 0.147) (Table 3 S4).

239 Success probabilities were time and state dependent (Figure 2). Mean success probabilities were 0.544 ±

240 0.253 for successful breeders, 0.496 ± 0.213 for failed breeders and 0.489 ± 0.300 for non-breeders.

241 Breeding success probabilities showed significant positive linear trends for successful breeders (slope =

242 0.501 ± 0.098) and failed breeders (slope = 0.363 ± 0.112) (Table 3 S4).

244 <u>Population modelling</u>

The stochastic λ obtained from the estimated demographic parameters (average rates for the entire 32-year

246 period) was 0.967 ± 0.062 and did not allow maintenance of the population.

247 Retrospective analyses indicated that survival of breeding individuals were the demographic parameters that

contributed the most in the variability in λ, followed by breeding success of successful individuals and

survival of non-breeders (Table 2). Elasticity of the population growth rate was the highest for adult survival,

then to juvenile survival, to recruitment at seven years old, to breeding probabilities and to breeding success

of breeders (Table 2).

252 Among the six scenarios combining conservation measures, population modelling indicated that when trawl

fisheries operated and no rat control occurred [scenario (1), 1986-1993], and when longline fisheries

operated without mitigation measures but rat control occurred [scenario (3), 1994-2004], stochastic λ did

not allow maintenance of the population (λ = 0.925 ± 0.071 and 0.954 ± 0.060 respectively).

When no rat control occurred, with or without implementation of bycatch mitigation measures in longline

257 fisheries [scenario (4) and scenario (2) respectively, see Appendix S5, Figure 1 S5], the population growth

258 rates were λ = 0.959 ± 0.071 and 0.914 ± 0.069 respectively (Figure 3).

259 When longline fisheries with mitigation measures operated and rat control occurred [scenario (5), 2005-

260 2016], the population slightly increased (λ = 1.014 ± 0.062) (Figure 3). The scenario with the highest λ was

261 the scenario where breeders' survival was modelled with no additive effect of longline fishery bycatch

262 [scenario (6), $\lambda = 1.035 \pm 0.062$, Figure 3].

263

264 <u>Effect of covariates and environmental sensitivity</u>

265 There was a significant negative effect of demersal fisheries effort in Crozet EEZ on annual survival

266 probability (slope = -0.512 $\pm$ 0.132, Figures 4 and 5). There was no significant effect of the climatic

267 covariates on survival.

268 There was evidence for a positive linear effect of SIC in the foraging areas during chick rearing on breeding

269 success of successful breeders (slope = 0.311 ± 0.073, Figure S8, Table S8), but no effect of detrended SIC

270 was detected. A positive linear effect of SSTa in the wintering area and of SSTa in the breeding area was

271 detected on breeding success of failed breeders (slope = 0.352 ± 0.092) and of successful breeders (slope =

272 0.333 ± 0.084) respectively, as well as a negative linear effect of SSTa and of SST in Antarctic ice front

273 area on breeding success of successful breeders (slope = -0.395 $\pm$ 0.100 and slope = -0.31 $\pm$ 0.076

274 respectively, Figure S8, Table S8). There was also evidence for quadratic effects of SOI at t-1 on survival

275 of successful breeders and of SIC on breeding success of successful breeders (Figure S8, Table S8).

276 Population growth rate was the most sensitive to demersal longline fisheries effort around Crozet (Table

277 S9), followed by SIC, SST and SSTa, to which it was between 1.6 and 3.2 less sensitive (Table S9).

278

Discussion

280 While considerable efforts were made for the last two decades in the implementation of bycatch mitigation

281 measures, this study is one of the few that assessed their effects on the demography and dynamic of a

282 threatened population. Our results clearly suggest a positive effect of mitigation measures with i) an increase

283 of the demographic parameter estimates after the implementation of bycatch mitigation measures, ii) a

284 decrease followed by an increase and stabilization of breeding densities before and after implementation,

285 respectively. Results indicated a higher sensitivity of population growth rate to demersal longline fishing

13

286 effort compared to climate covariates. However, we found that control of invasive predators was also

287 necessary in addition to bycatch mitigation measures to allow long-term persistence of the population.

288

289 <u>Role of bycatch mitigation measures on population dynamics</u>

290 As expected and following implementation of bycatch mitigation, in 2011 the white-chinned petrel breeding

291 population of Pointe Basse seemed to have partly recovered its density of 1983. Despite large confidence

292 intervals, this regain was still detected in 2017 and corresponded to an increase of 38% of the breeding

293 population density between 2004 and 2017. A similar increase was found in populations of another highly

294 accidentally caught large pelagic marine species, the porpoise (*Phocoena phocoena*) in California since the

295 limitation of bycatch (Forney et al., 2020). Pointe Basse being the largest breeding colony of white-chinned

296 petrels of Possession Island, we are convinced that this increase reflected the trend for the entire island

297 breeding population. Supporting this and in spite of large confidence intervals of population growth rates

298 that may assume stationary populations in all scenarios, population modelling indicated an increase of the

299 population growth rate at the study colony of Station de Pompage from the scenario (3) to the scenario (5),

300 that coincided with an increase in survival probability (from 0.87 to 0.91) and in breeding success

301 probability (from 0.58 to 0.69) from the period 1995-2004 to the period 2005-2017. This corresponded to a

302 change in longline fishery practices when bycatch mitigation measures were strongly reinforced in the

303 French EEZ. However, our matrix population models did not include immigration which could not be

304 estimated due to a lack of data. Thus we cannot exclude that the slight discrepancy between the observed

305 population growth rate estimated from breeding densities between 2004 and 2011 (1.090) and the population

306 growth rate obtained from scenario 6 (1.035) was partly due to immigration of white-chinned petrels to

307 Pointe Basse.

308

309 In bycatch studies, much attention was focused on survival rates of threatened populations (Gianuca et al.,

310 2019; Lewison & Crowder, 2003), because of the direct effect of bycatch on survival by additive mortality

311 (Barbraud et al., 2008), which can have a major influence on population trends because of the strong

312   sensitivity of the populations growth rate to this parameter in long-lived species (Gianuca et al., 2019;

313   Rolland et al., 2009; Jenouvrier et al., 2005). Our results are supportive of these statements since i) the

314   highest survival probabilities of the white-chinned petrel population were observed after the implementation

315   of bycatch mitigation measures, ii) this led to an increasing population, iii) population growth rate was more

316   sensitive to demersal longline fishing effort than to climate covariates which impacted breeding parameters.

317   Moreover, assessing the relationship between demersal longline fisheries effort and survival probability

318   allowed us to estimate survival with no additive mortality due to fisheries bycatch (0.923 ± 0.012). This was

319   very close to the survival probability estimated during the period with bycatch mitigation measures (2005-

320   2017), but also to the survival of other closely related Procellariform species (Gianuca et al., 2019; Oro et

321   al., 2004). Strong site and mate fidelity in this population (Bried & Jouventin, 1999) and no evidence of ring

322   loss suggest that adult survival estimates were poorly biased due to emigration.

323   Although we did not find any significant relationship between breeding probability and fishing effort

324   covariates, we cannot exclude that bycatch mitigation measures positively affected these demographic

325   parameters and contributed to explain part of their increases between the periods 1995-2004 and 2005-2017.

326   Indeed, bycatch may accentuate breeding failures in species with bi-parental care and may also decrease

327   breeding probability due to widowing (Mills & Ryan, 2005).

328

329   <u>Fisheries bycatch, rat predation and climate as drivers of population dynamics</u>

330   According to our results and to Barbraud et al. (2008), the white-chinned petrel population from Possession

331   Island declined when trawl fisheries (1986-1993, scenario (1)) and longline fisheries (1994-2004, scenario

332   (2)) were operating in the Crozet EEZ without or with partial mitigation measures. Our results suggest that

333   fisheries bycatch may have been the main driver of this decline, such as it was a serious threat for several

334   albatross populations (Lewison & Crowder, 2003; Pardo et al., 2017) and sea turtles (Lewison et al., 2004).

335   Indeed, we found a negative relationship between survival and longline fishing effort and low survival

336   probability prior to the implementation of fisheries bycatch mitigation measures. The population growth

337   rate was most sensitive to this covariate, and population modelling indicated an increase in population

338   growth rate when mitigation measures were implemented. In the Crozet EEZ, toothfish longline fisheries

339   appeared in the mid-1990s and ended trawling fisheries. At that time there was no mitigation measure for

340   seabird bycatch and IUU fisheries operated intensively until the early 2000s (Österblom & Sumaila, 2011).

341   During the 1980s, tuna fishing effort increased in the South Indian Ocean and could also have affected the

342   white-chinned petrel population [we observed the lowest survival probability (0.47) in 1991], as has been

343   observed for albatross populations (Tuck et al. 2003; Nel et al., 2002). Low survival and breeding success

344   probabilities estimated around 2000 may also be due to poor food availability and high fishing effort like it

345   was found on albatross species (Pardo et al., 2017). Those major changes in fishing practices (Duhamel et

346   al., 2011) that could have had contrasted effects on seabird populations and other large marine predators,

347   and the lack of reliable fishing effort data from the RFMOs, in particular in tuna fishery practices (Gilman,

348   2011; Rolland et al., 2009), could partly hinder the complete role of fisheries bycatch in population declines.

349   The improvement and stabilisation of the white-chinned petrel breeding success was probably largely due

350   to rat control in the study colony that occurred since 1994 (Jouventin et al., 2003). We found an increase

351   from 21% to 58% in success probability after the implementation of rat control and before the

352   implementation of fishery bycatch mitigation measures. Such an impact of rat predation on breeding success

353   suggests that the white-chinned petrel population may be stable without rat control but could hardly increase,

354   whether bycatch mitigation measures were implemented or not. As a consequence all population scenarios

355   (3, 5, 6) in which the effect of rat control was taken into account lead to an increase in population growth

356   rate compared to scenarios in which this effect was not accounted for. Rat control may have also benefited

357   to the breeding probability since a high breeding success could favour site and partner fidelity (Bried &

358   Jouventin, 1999).

359   Only breeding success was found to be affected by climate covariates, and the sensitivity of population

360   growth rate to these covariates was lower than sensitivity to longline fishing effort. Our results suggest that

361   the increase in breeding success during the study period was also likely due the increase of SIC from the

16

362 early 2000s. Although variations in SSTa affected breeding success, there was no temporal trend for these

363 covariates.

364

365 <u>Conservation implications</u>

366 Showing improvement in a large pelagic marine population following the implementation of fisheries

367 bycatch mitigation measures is encouraging. Such results highlight not only the efficacy of these measures,

368 but how those populations recover after a relatively important decrease. Our results showed that after the

369 control of rat predation, the implementation of bycatch mitigation measures was crucial for population

370 maintenance and is likely to become more and more important with the rise of new threats such as climate

371 change, diseases, or pollution. We view this study as an achievement of the significant efforts employed in

372 the conservation of albatrosses and large petrels threatened by fisheries bycatch. This highlights the

373 responsibilities that countries and RFMOs share for the conservation and management of endangered

374 species (Beal et al., 2021). We hope that it will encourage not only the managers to maintain and reinforce

375 protection measures, but also the scientific community to evaluate the demographic and population

376 consequences of such measures.

377 Rats were introduced in many islands (Jones et al., 2008) and their presence is a well-known threat to

378 seabirds (Croxall et al., 2012), including white-chinned petrels. Based on our results, we recommend rat

379 eradication from Possession Island to allow the long-term persistence of this petrel population. Considering

380 that different species may be impacted by the same threats but with different intensities, efforts in

381 conservation should be important in eradicating rat predation simultaneously with serious fisheries bycatch

382 mitigation management. Our adult survival estimate obtained with no additive mortality due to fisheries

383 bycatch remains slightly higher than the average survival during the period 2005-2017. This may suggest

384 that there is still some additional mortality due to bycatch within the at sea distribution of this white-chinned

385 petrel population. Recent reports from the Crozet EEZ indicate extremely low bycatch mortality in legal

386 longline fisheries [≤ 6 white-chinned petrels per year in average, (CCAMLR, 2020)]. Therefore, it is likely

17

387   that additive mortality is still occurring in international waters where efforts are needed to implement

388   effective mitigation measures.

389

Authors' contributions

391   All the authors conceived the ideas and designed methodology. KD, AC: data acquisition and preparation;

392   AD, CB : data analysis and processing, led the writing of the manuscript. All authors edited and revised the

393   manuscript and approved the final version. None of the author has conflict of interest related to this study.

Data availability statement

404   Data are available online at the Dryad Digital Repository: https://doi.org/10.5061/dryad.1zcrjdfts

405   (Dasnon et al, 2022)

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595 **Tables**

596 Table 1: Mean demographic parameters (*s*: survival, β: breeding, γ: success) white-chinned petrels at

597 Possession Island and associated standard errors according to breeding state (SB: successful breeder, FB:

598 failed breeder) and to the selected study period.

Demographic parameter	State	All period	1986-1993	1994-2004	2005-2017
s	SB	0.870 ± 0.099	0.800 ± 0.158	0.871 ± 0.045	0.910 ± 0.053
	FB	0.844 ± 0.106	0.881 ± 0.059	0.810 ± 0.109	0.853 ± 0.105
	NB	0.904 ± 0.017			
β	SB	0.870 ± 0.149	0.881 ± 0.094	0.802 ± 0.207	0.922 ± 0.058
	FB	0.825 ± 0.147	0.772 ± 0.202	0.832 ± 0.123	0.847 ± 0.058
	NB	0.350 ± 0.253	0.443 ± 0.302	0.256 ± 0.198	0.380 ± 0.250
γ	SB	0.544 ± 0.253	0.214 ± 0.224	0.579 ± 0.186	0.693 ± 0.113
	FB	0.496 ± 0.213	0.323 ± 0.199	0.546 ± 0.263	0.546 ± 0.101
	NB	0.489 ± 0.300	0.555 ± 0.248	0.536 ± 0.295	0.293 ± 0.178

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601 Table 2: Population growth rate sensitivities and elasticities to demographic parameters, and contribution
602 of demographic parameters to the variance (Var(λ) = 0.004) in population growth rate for white-chinned
603 petrels at Possession Island, 1987-2017. SB: successful breeder, FB: failed breeder, NB: non-breeder, j_n:
604 juvenile of n years old.

Parameter	Status	Estimate	Temporal variance	Elasticity	Sensitivity	Contribution (%)
Juvenile survival	j_1	0.7		0.068	0.094	
	j_2	0.761		0.068	0.086	
	j_3	0.796		0.068	0.083	
	j_4	0.821		0.068	0.080	
	j_5	0.841		0.068	0.078	
	j_6	0.857		0.063	0.071	
	j_7	0.870		0.042	0.046	
Recruitment	j_5	0.062		<0.001	0.014	
	j_6	0.314		0.002	0.007	
	j_7	1		0.042	0.040	
Survival	SB	0.870	0.0095	0.200	0.223	28.0
	FB	0.844	0.0108	0.181	0.201	26.2
	NB	0.844	0.0108	0.108	0.126	10.1
Breeding	SB	0.870	0.0216	0.059	0.066	5.4
	FB	0.825	0.0204	0.049	0.062	4.8
	NB	0.350	0.0635	0.015	0.041	6.5
Success	SB	0.544	0.0635	0.031	0.056	11.9
	FB	0.496	0.0445	0.025	0.047	6.0
	NB	0.489	0.0835	0.008	0.016	1.2

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