# Demography of the critically endangered Balearic shearwater: the impact of fisheries and time to extinction

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Running title: Impact of fisheries on Balearic shearwaters

**Summary**

1. Worldwide, many seabirds are affected by fisheries in opposing ways: as a source of mortality from bycatch, but also by providing discards as a predictable and abundant food resource. This applies to the Balearic shearwater *Puffinus mauretanicus*, the most endangered European seabird, whose time to extinction was estimated at only ~40 years a decade ago.

2. Since the previous assessment, new data and more sophisticated demographic modelling have become available, and new fishing policies from the European Union (Common Fisheries Policy, CFP) will apply, posing different scenarios for the viability of the species. Thus, there is both an urgent need and an opportunity for a more reliable update of the conservation status of the species.

3. Demographic data were collected between 1985–2014 at one of the world's largest colonies. Most demographic parameters were estimated using multi-event capture–recapture modelling. Some parameters, such as bycatch rate, immature individual survival and recruitment were estimated for the first time. We incorporated estimates into stochastic population models to forecast time to extinction and assess the viability under different management scenarios, accounting for upcoming fishing policies.

4. Adult survival was much lower than expected (0.809, SE: 0.013), and largely influenced by bycatch, which accounted for a minimum of 0.455 (SE: 0.230) of total mortality. Breeding success was positively correlated with discard availability. Recruitment started at low rates in 3-year-old birds (0.030, SE: 0.0455), increasing in following age classes and was almost complete at 6 years.

Under the present scenario, we predict a time to extinction of 61 years (95% CI: 55–69).

5. *Synthesis and applications*. Population projections suggest that the actual impact of fisheries on Balearic shearwaters is unsustainable and the imminent discard bans under the new Common Fisheries Policy may accelerate the declining trend. This study demonstrates that reducing the bycatch rates of fisheries is an unavoidable and urgent conservation measure for avoiding the extinction of the species. We also advise setting up demographic long-term studies, to allow researchers to diagnose, with reliability, the effectiveness of management actions. These actions will also benefit many other marine top-predator species affected by this anthropogenic impact.

**Key-words**: bycatch, conservation, discards, environmental policies, extinction, marine predator, multi-event capture–recapture, population models, survival

## Introduction

Effective management of species, especially for threatened taxa, should begin by estimating, with accuracy, their demographic parameters in order to assess population growth rate, elaborating a conservation diagnosis and making predictions about population (Caswell 2001; Morris & Doak 2002). More importantly, conservation-oriented science informs managers about what demographic parameters are unsustainable, aiding optimization of priority actions and ensuring that efforts are concentrated on those parameters that most need to be remedied (e.g. Norris 2004). This is essential when dealing with endangered species, because time to reverse sharp declining trends is limited and good management should target the most effective

actions. Nevertheless, many endangered species are neither abundant nor widespread, and surveillance monitoring typically provides weak inferences about population declines.

This is the case for the Balearic shearwater *Puffinus mauretanicus*, the most threatened seabird in Europe and listed as critically endangered, which is the highest IUCN threat category for a taxon in the wild (BirdLife International 2015). Balearic shearwaters breed mostly in inaccessible sites, so it is unfeasible to monitor population trends using counts, abundance indices or similar techniques. Categorization of the species was made 10 years ago using scattered demographic information applied to a population viability analysis (PVA) that predicted a mean extinction time of only ~40 years (Oro *et al.* 2004). Since that assessment, new demographic and population data have become available, more sophisticated demographic modelling has been developed, and new fisheries policies from the European Union have been introduced (Common Fisheries Policy: CFP); these changes pose a different future scenario for the Balearic shearwater and highlight the urgent need for a more reliable update of its conservation status.

The census of the breeding population was recently updated to ~3200 pairs, a much larger figure than the previous estimate of 2000 pairs (Arcos 2011). Recent at-sea and coastal-based surveys suggest a global population in the range of 20 000–30 000 individuals (Arcos *et al.* 2012). These new figures make necessary a reassessment of the conservation status of the species, but to do it properly survival estimates should also be updated, and bycatch mortality quantified (Oro *et al.* 2004; ICES 2013). Incidental capture or bycatch represents ~8% of global fisheries production (Kelleher 2005), and is a major threat for seabirds, particularly for the Procellariiformes (Anderson *et al.* 2011, Croxall *et al.* 2012), and the Balearic shearwater is no exception (Cooper *et al.* 2003; Laneri *et al.* 2010; ICES 2013). Conversely, there is also evidence that Balearic

shearwaters reap some benefit from fisheries through discard scavenging, e.g. Arcos & Oro (2002) found that >40% of the energy requirements of Balearic shearwaters during the breeding season came from trawler discards. The increase in food availability provided by discards (and their high predictability in space and time) seems to be responsible for the growth of many seabird populations, mediated by a positive effect on reproductive success and probably also survival (Oro *et al.* 2013). However, forthcoming CFP regulations, aimed at banning discard practices, could negatively influence the breeding performance of this species, at least in the short term (Bicknell *et al.* 2013). More positively, bycatch mortality is expected to decrease in the future, as increasing awareness leads to mitigation action through a specific Plan of Action (see Table S1 in Supporting Information).

We used historical ringing and recovery data collected between 1985 and 2014, and new at-sea estimates of population size to (i) update adult survival probability and estimate for the first time: immature individual survival, recruitment probability by age, and the probability of dying in fishing gear, by using multi-event capture–recapture modelling; (ii) estimate breeding success and assess the influence of fishing discards on this parameter; and (iii) construct population models for assessing the probability of extinction under several scenarios considering the contrasting effects of fisheries. While Oro *et al.* (2004) used a limited number of years to estimate demographic parameters, the present study uses recently developed capture–recapture models that allowed us to exploit a much larger data set covering 30 years of monitoring.

## Materials and methods

### STUDY AREA AND FIELD METHODS

Data were collected at Sa Cella cave (on the north-west coast of Mallorca, Balearic Islands, Spain), which had one of the largest colonies of the species (~170–200 breeding pairs) between 1985 and 2014, except for the period 2005–2009, when the colony was not monitored. The colony is free of carnivores and rats, and legally protected.

Adults and chicks were trapped by hand, marked with stainless steel rings with a unique code (see details in Oro *et al.* 2004) and their breeding status assigned (either breeder or unknown). Recoveries were obtained from the Spanish ringing office (SEO/BirdLife) and provided by fishermen, researchers and wildlife recovery centres. Each recovery was assigned as caused by longline fishing (carrying a hook) or unknown.

## ANALYSIS OF DEMOGRAPHIC PARAMETERS

To estimate survival, recruitment and probability of dying in fishing gear, mark–recapture and recovery data were jointly analysed in the frame of multi-event capture–recapture modelling (Pradel 2005). In this modelling approach we relate the true states of the individuals with the observed events through a series of conditional probabilities (Pradel 2005). Models were fitted in program E-SURGE (Choquet, Rouan & Pradel 2009), which distinguishes three basic types of parameters: the initial state probabilities, the transition probabilities between states, and the event probabilities. Model selection relied on QAICc, i.e. the Akaike Information Criterion corrected for overdispersion and for small sample sizes (Burnham & Anderson 2002). Because there is no goodness-of-fit test available for multi-event models, we assessed the fit of a model that only retains whether an individual is encountered or not (Cormack-Jolly-Seber type models) using U-care (Choquet *et al.* 2009).

146 Given that the capture–recapture effort was not uniform during all study years, we
147 initially performed an analysis with a reduced data set containing no data collection
148 gaps, to extract reliable estimates of age at recruitment (*Recruitment analysis*). We
149 subsequently carried out a second analysis on the complete data set, and fixed age of
150 recruitment to estimate both immature and adult survival, and the probability of
151 fisheries-related mortality (*Global analysis*). We distinguished between breeder and
152 non-breeder survival, because those not observed as breeders may be transients, and
153 therefore artificially reduce estimated local survival rates. Our model incorporated an
154 error probability of ascertaining an individual's breeding status, i.e. the probability that
155 a breeder was not observed to be breeding.

156 *Recruitment analysis*

157 Using data from 1994–2004, we classified individuals into two groups based on the age
158 at first capture (chicks and adults). Models included three biological states: alive
159 breeder (B), alive non-breeder (NB) and dead (D). The last state was not observable and
160 the initial state in our models was always NB in animals marked as chicks. Transitions
161 between states were modelled in a two-step approach: survival and recapture
162 probability. In each capture–recapture occasion ('*t*') we considered three possible
163 events: individual not seen (noted 0), individual seen alive but with unknown breeding
164 status (noted 1) and individual seen breeding (noted 2). Recruitment was defined as the
165 probability $r_i$ of breeding for the first time at each age $i$ and equalled the transition from
166 the state of non-breeder to breeder. As birds do not visit the colony before 3 years of age
167 (Oro *et al.* 2004, own data), survival was modelled separately for both immature
168 individuals (1- and 2-year-old) and adults (for breeders and non-breeders), and was kept
169 constant in all models (see details in Appendix S1). We also undertook a model run to
170 test whether survival of immature individuals was equivalent to non-breeding adults.

171 Then, we tested several models considering different age curves at recruitment, starting
172 at 3-years-old and with full recruitment from 3 to ≥7 years old. We had no power to test
173 for longer age curves of recruitment, although results suggested that recruitment at older
174 ages was likely rare.

175 *Global analysis*

176 We analysed all data available from 1985 to 2014 and, as previously, individuals were
177 classified in two groups based on their age at first capture. Models included five
178 biological states: alive breeder (B); alive non-breeder (NB); individual recently dead by
179 longline bycatch (RF); individual recently dead by unknown causes (RD); and dead (D),
180 this last state being non-observable. The initial state in our models was always NB for
181 chicks. Transitions between states were modelled in a three-step approach: survival,
182 probability of death in bycatch events and recruitment probability. In each capture–
183 recapture occasion ('*t*') we considered five possible events: Individual not seen (noted
184 0); individual seen alive but with unknown breeding status (noted 1); individual seen
185 breeding (noted 2); individual found recently dead by bycatch (noted 3); individual
186 found recently dead by unknown causes (noted 4) (see details in Appendix S1).

187 As in the Recruitment analysis, we assumed two different survivals: immature and
188 adult, the latter considering different survival for breeders and non-breeders. Given that
189 only ringing (but not recapture) was carried out from 1985 to 1996, we estimated
190 survival separately for the two periods (1985–1996 and 1997–2014), and assumed only
191 estimates from the second period were reliable. We additionally undertook one model
192 run assuming the same survival for the whole study period, to be more confident in our
193 assumptions. We also tested for a time variant survival for immature individuals and
194 adults. We estimated the probability of dying in fishing gear, conditional to dying, and

195 we additionally tested an age effect on this probability, i.e. separately for immature
196 individuals and adults. After modelling first recapture probabilities (models not shown
197 except model 5 in Table 1), we selected a model in which recapture probabilities were
198 kept time-variant except for the last five years of the study, when fieldwork effort was
199 constant. As we had not enough data to check if recovery probability varied over time,
200 we kept it constant. Additionally, due to our limited recovery data sample size, we
201 assumed equal recovery rates for bycatch individuals and those dead by unknown
202 causes. We should state that even though this may not be advisable (Schaub & Pradel
203 2002), here both recovery rates are very small and even if some bias does exist, it is not
204 clear in which direction and it is probably not large (see also results from *Bycatch*
205 *events in Balearic shearwaters* section below).

206 Once the probability of bycatch was estimated, we then roughly estimated hypothetical
207 survival without incidental capture, both for immature individuals and adults. We did so
208 by adding the estimated probability of dying in fishing gear to the survival probability,
209 assuming additive mortality in both age classes.

210 *Breeding success and fishing discards*

211 Sant Carles de la Ràpita harbour holds the bulk of the important trawling fleet operating
212 off the Ebro Delta, where Balearic shearwaters often forage (Louzao *et al.* 2006). The
213 amount of trawling discards and trawling landings are correlated (Oro & Ruiz 1997),
214 thus we used the statistics of trawling landings at this harbour between March and June
215 (i.e. encompassing most of the breeding cycle) as a proxy for interannual variability in
216 food availability. Breeding success of monitored study nests was calculated between
217 1997–2004 and 2010–2013, as the number of fledglings by eggs laid, each season. We
218 then used generalized linear models (GLM), with a logit link function and binomial

219 error, to test for the potential association between our proxy of food availability and
220 breeding success over the 12-year period. The intercept of this logistic regression
221 function corresponded to the estimated breeding success in the absence of discards, and
222 this value was used as the breeding success in the scenarios with discard banning.

223 *Bycatch events in Balearic shearwaters*

224 In addition to the capture–recapture analyses we also gathered additional information on
225 registered bycatch events in Balearic shearwaters. Data were obtained from the
226 literature and from Wildlife Recovery Centres in Catalonia, Valence and Balearic
227 Islands, where the species mostly ranges. We gathered data from all Balearic
228 shearwaters entering one of these Wildlife Recovery centres from 1985 to 2014. For
229 each animal arriving at the centre, the cause of death was recorded, if possible.

230 POPULATION MODELLING

231 We formulated a seven stage-class matrix population model (Fig. 1, Table S2) to assess
232 the population growth rate of Balearic shearwaters under current and possible future
233 environmental conditions. The model followed a pre-breeding census format, and was
234 based only on females; assuming equal survival between sexes and monogamy (Oro *et*
235 *al.* 2004). All projection models were developed and executed in program R
236 (http://cran.r-project.org).

237 *Deterministic analysis*

238 We first carried out a deterministic analysis that included mean values of the estimated
239 vital rates and yielded the deterministic population growth rate or λ (largest eigenvalue
240 of the population matrix, Caswell (2001)). All the vital rates used in the model were
241 derived from this study, except the probability of skipping breeding, which we obtained

242 from Oro *et al.* (2004). To initialize the models we used the highest available estimate
243 of current population size, obtained from at-sea censuses (Arcos *et al.* 2012; Arroyo *et*
244 *al.* 2014). In addition to population growth rates, the deterministic model was used to
245 estimate other important information, such as the stable age distribution, generation
246 time, reproductive value and sensitivities and elasticities. We then used the estimated
247 stable age distribution to initialize the stochastic models.

248 *Stochastic analysis*

249 While the deterministic growth rate describes the population trend for constant,
250 invariant vital rates, we also constructed stochastic models and we account for
251 parameter uncertainty and annual variability in those rates to assess the risk of
252 population decline or extinction. To do so, we first assessed temporal variability in
253 survival parameters in the species. Based on the time varying model, we decomposed
254 variance and found that all the observed variance was due to sampling variance;
255 additionally, the model assuming annual variability in survival performed worse than
256 those assuming constant survival (Model 4 in Table 1) thus we just included parameter
257 uncertainty in survival parameters in our models, but not temporal stochasticity. We
258 also assessed temporal variability in fecundity by estimating the process variance in this
259 parameter and then used this variance to account for stochasticity in fertility in our
260 projections. As we could not estimate temporal variability in recruitment, or skipping
261 probability, we left these parameters constant over time and only took into account
262 parameter uncertainty in our projections. To include temporal stochasticity in fecundity
263 we picked random values from beta distributions in each year of simulations, using the
264 mean and process variance values. We did not consider density-dependence in our
265 model because population growth rate was negative in all cases. Models were run using
266 Monte Carlo simulations for '100 years' and '1000 population' trajectories. We ran

models under different scenarios considering the current fisheries impact, and hypothetical scenarios with different combinations of bycatch intensity and discard availability according to EU fishing policies (Table 2). We also set some scenarios using the lowest survival estimates from a range of published values for similar Procellariiformes of the *Puffinus* genus, which are less affected by bycatch and other anthropogenic mortalities (Table S3). In all scenarios, survival of non-breeders was considered to be equal to the survival of breeders, because we assumed that environmental stochasticity equally affected the two groups (Table 2). Under all scenarios we estimated the mean stochastic population growth rate ($\lambda_s$) over a short and relevant time horizon of 100 years from 1000 projections, together with 95% confidence intervals:

$$\lambda_s = \frac{1}{1000} \sum_{i=1}^{1000} \exp\left[ \frac{\ln(N_i(T=100)) - \ln(N_i(T=0))}{100} \right]$$

*Detecting overharvesting*

We further evaluated the impact of longline bycatch as an additional source of mortality, using the "potential biological removal" PBR (Dillingham & Fletcher 2008). We first calculated the maximum potential annual growth rate ($\lambda_{max}$) by means of the "demographic invariant method" DIM (Niel & Lebreton 2005):

$$\lambda_{\max} \approx \frac{(s\alpha - s + \alpha + 1) + \sqrt{(s - s\alpha - \alpha - 1)^2 - 4s\alpha^2}}{2\alpha},$$

which assumes constant adult survival probability $s$ and the average age at first reproduction $\alpha$. Since $s$ was affected by longline bycatch (see Results), we took the average minimum survival estimates from studies on closely related Procellariiformes

not affected by additive mortality (0.917, see Table S3). To obtain $\alpha$ we first calculate $\alpha_i$ (the probability of a bird of age $i$ being a first-time breeder) from our recruitment probability $r_i$ through the equation:

$$\alpha_i = r_i \prod_{y \le j < i} (1 - r_j), i \le f ,$$

where $y$ was the youngest age at breeding and $f$ was the full age at recruitment (Pradel & Lebreton 1999). From $\alpha_i$ we obtained $\alpha$ as:

$$\alpha = \sum_i \alpha_i ,$$

which equalled 4.83 for Balearic shearwaters in our study.

Then we calculated PBR as:

$$PBR = \frac{1}{2} R_{\max} N_{\min} f$$

where $R_{\max}$ is the maximum annual recruitment rate, equalling ($\lambda_{\max}$ - 1), $N_{\min}$ is a conservative estimate of population size and $f$ is a recovery factor with values ranging from 0.1 to 1 depending on population conservation status and the best adaptive management action to be taken.

To calculate $N_{\min}$, we took the 20$^{th}$ percentile of the distribution of population size following the equation (Dillingham & Fletcher 2008):

$$N_{\min} = \hat{N} \exp\left(-0.84\sqrt{\ln(1 + CV_N^2)}\right),$$

where $\hat{N}$ equals 23 780 individuals, and $CV_N$ equals 0.03, using mean and its 95% CI of that estimate provided by Arroyo *et al.* (2014). We set $f$ at a conservative value of 0.1, typical for endangered species.

## Results

During March–June of 1985–2014 a total of 1344 individuals were captured and ringed at the study colony, corresponding to 761 chicks (57%) and 583 adults (43%). A total of 394 marked individuals were recaptured at least once, of which 179 were marked as chicks and recruited as breeders at the study colony (24% of all ringed chicks). More than half of the marked adults (54%) were never recaptured. We obtained 11 recoveries, five dead from bycatch and six from unknown causes.

*Recruitment analysis*

The goodness-of-fit for the Cormack-Jolly-Seber model was poor ($\hat{c}$ =3.993) mainly due to a transient effect from individuals ringed as chicks. Thus, we included age in our models and then corrected for the remaining overdispersion with a $\hat{c}$ = 2.270.

Three models were best ranked in model selection: suggesting that there are two, three or four ages of recruitment (from 3 to 6, Models 1, 2 and 3 in Table 3), respectively. Estimates from the three models were very close (Table S4), and showed that most individuals were recruited by 6-years-old, with low recruitment at 3 years of age. We took recruitment estimates from the intermediate Model 2 for assessing population viability. Probabilities of recruitment at age *i* ($r_i$; mean and SE) from Model 2 were: $r_3$ = 0.030 (SE: 0.045), $r_4$ = 0.243 (SE: 0.122), $r_{>5}$ = 0.431 (SE: 0.155). The model assuming equal survival for immature individuals and non-breeding adults was not well supported (Model 6, Table 3).

*Global analysis*

When analysing the complete data set, the goodness-of-fit for the Cormack-Jolly-Seber model was poor ($\hat{c}$ = 3.264) due to the presence of transients among individuals ringed as chicks. We included age in our models and corrected for remaining overdispersion with a $\hat{c}$ = 2.350.

The model with the lowest QAICc value (Model 1, Table 1; parameters estimates on Table S5) differentiated the two periods with and without recaptures: this model indicated that survival did not vary significantly over the years and it was much lower for 1- and 2-year-old individuals (immature) than for older birds: 0.434 (95% CI: 0.351–0.520) and 0.809 (95% CI: 0.782–0.833), respectively. Survival of non-breeders was not estimable given our data. The model considering the whole study period (1985–2014) had a higher QAICc value (Model 3, Table 1) and confirmed that the first period without recaptures was only valuable for using birds marked during this period. Given the limited data on bycatch events, we could not disentangle if there was a different bycatch probability for immature individuals and adults, because both models had similar QAICc values (Models 1 and 2, Table 1). Incidental capture in longlines was estimated at 0.455 (95% CI: 0.119–0.837), which meant that approximately half of mortality was attributable to bycatch, with a probability of mortality from longlines of 0.256 and 0.087 in immature individuals and adults, respectively. Hypothetical local survival without incidental capture was thus estimated at 0.691 (SE: 0.043) and 0.896 (SE: 0.013) for immature individuals and adults, respectively. We should notice that the confidence interval for bycatch probability is extremely high due to the limited sample size on recovery data, so these estimates should be treated with caution.

*Breeding success and fishing discards*

355 Mean breeding success at the study colony was estimated at 0.665 (SE: 0.038), ranging
356 from 0.400 to 0.920 fledglings per breeding pair. Breeding success was positively
357 associated with trawling landings ($z$ = 3.170, d.f. = 11, $P$ = 0.001, Fig. S1). The
358 intercept of the logistic regression function corresponding to the estimated breeding
359 success in the absence of discards was 0.433 fledglings per pair (SE = 0.137).

360

361 *Bycatch events in Balearic shearwaters*

362 When gathering data from the literature we observed that bycatch events are frequent in
363 the species even if their occurrence seems difficult to quantify (Table S6).

364 From 1985 to 2014, we found registered deaths of 78 Balearic shearwaters at Wildlife
365 Recovery Centres in Valence, Catalonia and the Balearic Islands. In 57.7% of the cases
366 the cause of death could not be determined. From the remaining individuals (43.3%), in
367 more than the half of the cases (67%), the cause of death was bycatch in fisheries gear,
368 and longline bycatch represented 48.5% of the total known mortality.

369

370 POPULATION MODELLING

371 *Deterministic analysis*

372 The estimated deterministic λ was 0.855, reflecting an annual decline of about 14% in
373 population size, and a generation time of 12.8 years. The stable stage distribution for the
374 species showed that 57.3% of females are breeders (Table S7). Hence, taking into
375 account the recent global population estimate, the number of breeding pairs should be
376 ca. 7200. Sensitivity and elasticity analysis showed that changes in survival of breeding

adults, and to a smaller extent the probability of a skipping breeder to reproduce again, had the largest effect on the population growth rate, and fertility the smallest effect (Table S7).

*Stochastic analysis*

When adding environmental stochasticity under current conditions, the mean growth rate for the population $\lambda_s$ was 0.856 (95% CI: 0.841–0.872) (Table 2, Fig.2). We estimated a time to extinction of 61 years (95% CI: 55–69). The only scenarios with stable or increasing trends were those in which survival reached values comparable to those described for closely related Procellariiformes (scenarios 5 and 6, Table 2, Fig.2). With these higher survival probabilities, the population should avoid extinction even with a ban on discards reducing fertility (Scenario 5, Table 2).

*Detecting overharvesting*

Using the DIM approach, $\lambda_{max}$ was 1.101 (range 1.087–1.112), i.e. that under ideal demographic conditions, the population cannot grow at a rate higher than 11.2% per year. A conservative estimate of population size $N_{min}$ was calculated at 19 965 shearwaters, from which we estimated a PBR of 100 shearwaters dying in fishing gear each year (range 87–112).

## Discussion

Fossil records of Balearic shearwaters suggest that they had a very large population until the arrival of human colonizers to the Balearic archipelago ~$4.2\times10^3$ years ago, which brought alien carnivores and rodents that have decimated most of the breeding sites (Alcover, Seguí & Bover 1999). Harvesting was also a major pressure in historical

times, though it is residual nowadays. New anthropogenic impacts appeared in recent decades, notably habitat loss by urbanization and bycatch in fisheries (Table S1) (Lewison *et al.* 2012). Oro *et al*. (2004) performed a PVA using demographic data from two predator-free sites and concluded that the population would reach extinction in a few decades. Ten years later, our results confirm this prediction, despite considering a larger base population. Our latest results should be considered as more robust, as they are based on a larger (and updated) data set, and use improved, up-to-date capture–recapture modelling procedures. Moreover, they show that fisheries are a crucial factor for the viability of the species.

Under the present scenario we predicted a time to extinction of 61 years, which confirms that the Balearic shearwater is one of the most endangered bird species in the western Palaearctic (BirdLife International 2015). Two opposite biases may have occurred in our study. First, survival and fertility were probably overestimated, because these parameters are impacted in most colonies by alien predators (Arcos 2011), but were not present in the study colony. Second, our survival estimates were local, i.e. did not distinguish mortality from permanent dispersal. While this last bias was likely very small for adult survival (breeding dispersal in Procellariiformes is very low, e.g. Sanz-Aguilar *et al.* (2011)), it might be important for immature survival, since natal dispersal may not be negligible (Genovart *et al.* 2007). However, we should note that there may be immigration of pre-breeders from other colonies. We should also highlight that our estimate of bycatch rate is based on very limited recovery data and thus it contains much uncertainty and should be treat with caution. Additionally this estimate may be biased in either direction; bycaught birds may be more easily recovered than animals that have died from other causes, but also some recovered birds were assigned as having an unknown cause of death, but they may be bycaught birds, e.g. animals found on

beaches after drowning when released from fishing gear entanglement (Generalitat Valenciana 2012). However, this rough estimate is closely matched by the bycatch rate estimated from the Wildlife Recovery Centres' data. Overall, our prediction for the current scenario was thus rather conservative, indicating that urgent conservation action is necessary to halt the extinction of the Balearic shearwater.

## THE IMPACT OF FISHERIES

Incidental capture in fishing gear represents a major cause of additive mortality for many seabirds worldwide, and it has been the focus of conservation concern and research in the last three decades (Lewison *et al.* 2012). Observer on-board programmes for longline vessels in the Mediterranean have reported low rates of bycatch for Balearic shearwaters (Belda & Sanchez 2001; Laneri *et al.* 2010), although there is increasing evidence of regular mortality, particularly by demersal longlines (ICES 2013). Moreover, events of "mass" mortality, with over 100 birds per event, appear to occur with relative frequency, although they are difficult to detect through observer programmes with limited coverage (Besson 1973; Arcos, Louzao & Oro 2008; ICES 2008; Louzao *et al.* 2011). Bycatch impacts from other gear, such as trawlers and purse-seine vessels, have also been reported recently (Oliveira *et al.* 2015). Despite the fact that there is not a reliable estimate of the number of birds caught per year, there is no doubt that this figure is well above our estimated PBR value, and the rough estimated bycatch rate of about half of the mortality detected in Balearic shearwaters confirms that current fishery impact is unsustainable. The only scenarios yielding positive population growth rates were those assuming survival rates of other *Puffinus* species with little or no anthropogenic mortality.

By quantifying the bycatch mortality in Balearic shearwaters here, we demonstrate that the actual rate is not compatible with the viability of the species. We thus recommend the implementation of urgent mitigation actions to reduce fisheries bycatch rates in this and other top-predator species severely affected by this anthropogenic impact. The imminent scenario arising from EU fishing policies poses both threats and opportunities for many seabirds, and especially for the critically endangered Balearic shearwater. Seabird bycatch has been incorporated into the EU agenda, and efforts to reduce this source of mortality are expected (Table S1), although so far progress has been very slow (ICES 2013). Conversely, the so-called "discard ban" (Table S1), if ultimately beneficial for the marine ecosystem, could bring negative effects for the Balearic shearwater and other seabirds in the short term (Bicknell *et al.* 2013). First, it could accelerate the decline of the species by reducing breeding success. Second, attendance and bycatch risk of shearwaters at longline vessels and other fleets may increase when trawlers do not operate (Garcia-Barcelona *et al.* 2010; Laneri *et al.* 2010), so a discard ban might increase bycatch and thus extinction probabilities. In the long term, however, if the discard reduction is actually accompanied by efforts to increase selectivity and reduce fishing pressure, this should be regarded as a beneficial measure for the seabirds, as fish stocks (i.e. natural prey) are expected to recover.

## CONCLUSIONS AND RECOMMENDATIONS

Survival as well as bycatch mortality estimates, suggest that the global population of Balearic shearwaters is not viable in the long term. While the impact of alien predators can, and should be, urgently addressed (Nogales *et al.* 2004), actions to stop or reduce bycatch are fraught with challenges because of the large spatial scales to be covered (Guilford *et al.* 2012; Louzao *et al.* 2012), the range of multi-national fishing fleets

involved and socio-economic considerations. However, reducing bycatch rates in the short term is unavoidable and an urgent conservation priority. More data are required to determine which factors increase bycatch rates and which are the critical areas with highest impact, and it is crucial to then apply measures such as time restrictions on fishing activity, bycatch mitigation technology and practices, as well as the education of stakeholders and consumers. Finally, it is essential to set up demographic long-term studies, to allow researchers to diagnose with reliability the effectiveness of all those actions and to apply an adaptive management process (Lahoz-Monfort, Guillera-Arroita & Hauser 2014). Although this would require long-term financial investment, these studies would also be relevant to a wide range of seabirds and marine predators, as well as to the whole marine ecosystem.

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## Data accessibility

Balearic shearwater data are available http://cedai.imedea.uib-csic.es/geonetwork/srv/es/main.home?uuid=637a7431-3b19-43e4-acb5-d1a09534eaf9.

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**Table 1**. Model selection from the *global analysis* (see Materials and methods) for estimating survival and the probability of dying in fishing gear, by age (immature and adult). Recruitment probability was fixed at values previously estimated. Given that no resightings were carried out from 1985 to 1997, some models considered two separate periods: 1985–1996 and 1997–2014. Recapture probability was fixed to zero in years with no resightings. In all models except model 5, we left recapture probability to vary annually during the first 24 years of the study (1985–2009*t) and assumed to be constant during the last five years of the study (2010–2014), when the recapture effort was highly constant among years. Np = number of identifiable parameters. $w_i$ = Akaike weight, which represent the relative likelihood of model *i*

| Model | Survival | Bycatch | Recapture | Np | Deviance | QAICc | ΔQAICc | $w_i$ |
|---|---|---|---|---|---|---|---|---|
| 1 | Constant by age, two periods | Constant | 1985–2009*t, 2010–2014 | 34 | 4675.759 | 2055.399 | 0 | 0.634 |
| 2 | Constant by age, two periods | By age | 1985–2009*t, 2010–2014 | 35 | 4673.479 | 2056.495 | 1.096 | 0.366 |
| 3 | Constant by age, one period | Constant | 1985–2009*t, 2010–2014 | 31 | 4726.910 | 2070.945 | 15.546 | 0.000 |
| 4 | Age, Time varying, two periods | Constant | 1985–2009*t, 2010–2014 | 71 | 4600.788 | 2101.242 | 45.843 | 0.000 |
| 5 | Constant by age, two periods | Constant | 1985–2009*t, 2010–2014*t | 42 | 4669.172 | 2069.174 | 13.775 | 0.000 |

**Table 2**. Estimates of demographic parameters used in population models (standard errors in brackets) for each scenario considered, together with its mean stochastic population growth rate $\lambda_s$ and 95% confidence intervals. Scenario 1: current situation. Scenario 2: reduced breeding success under future ban of discards. Scenario 3: conditions under future ban of discards but bycatch reduced. Scenario 4: current situation and bycatch reduced. Scenario 5: hypothetical conditions with minimum survival probabilities described for closely related Procellariiformes in optimal environments, and with a ban of discards. Scenario 6: Same demographic parameters as scenario 5 but no ban of discards. Sex ratio was set to 0.5 in all models. Recruitment and sabbatical estimates were common for all scenarios; recruitment was 1 for individuals >6 years old

| Scenario | 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|
| Survival affected by | yes | yes | no | no | no | no |
| Discard banning | no | yes | yes | no | yes | no |
| Demographic parameter | | | | | | |
| Adult survival | 0.809 (0.013) | 0.809 (0.013) | 0.896 (0.013) | 0.896 (0.013) | 0.917 (0.014) | 0.917 (0.014) |
| Immature individual survival (1–2 years) | 0.434 (0.043) | 0.434 (0.043) | 0.691 (0.043) | 0.691 (0.043) | 0.853 (0.043) | 0.853 (0.043) |

| | | | | | | |
|---|---|---|---|---|---|---|
| Breeding success | 0.665 (0.134) | 0.433 (0.137) | 0.433 (0.137) | 0.665 (0.134) | 0.433 (0.137) | 0.665 (0.134) |
| Sabbatical probability | | | 0.261 (0.063) | | | |
| Recruitment probability | | | | | | |
| 3 years | | | 0.030 (0.045) | | | |
| 4 years | | | 0.243 (0.122) | | | |
| ≥ 5years | | | 0.431 (0.155) | | | |
| $\lambda_s$ | 0.856 | 0.848 | 0.951 | 0.972 | 1.006 | 1.044 |
| $\lambda_s$ lower 95% CI | 0.841 | 0.838 | 0.938 | 0.955 | 0.993 | 1.002 |
| $\lambda_s$ upper 95% CI | 0.872 | 0.860 | 0.965 | 0.989 | 1.020 | 1. 079 |

**Table 3**. Model selection for *recruitment analysis* (see Materials and methods). Notation for recruitment indicated the different age groups considered: for instance, "3, 4, ≥5" showed different recruitment probabilities for 3- and 4-years-old and older birds. Np = number of identifiable parameters. $w_i$ = Akaike weight, which represents the relative likelihood of model *i*.

| Model | Survival | Recruitment | Np | Deviance | QAICc | ΔQAICc | $w_i$ |
|---|---|---|---|---|---|---|---|
| 1 | Two age classes | 3, ≥4 | 25 | 2708.324 | 1244.584 | 0 | 0.340 |
| 2 | Two age classes | 3,4, ≥5 | 26 | 2706.118 | 1245.682 | 1.097 | 0.197 |
| 3 | Two age classes | 3,4,5, ≥6 | 27 | 2702.278 | 1246.061 | 1.477 | 0.160 |
| 4 | Two age classes | 3,4,5,6, ≥7 | 28 | 2702.015 | 1248.020 | 3.436 | 0.060 |
| 5 | Two age classes | constant | 24 | 2723.051 | 1249.009 | 4.425 | 0.040 |
| 6 | Breeders/Non Breeders | 3,4,5, ≥6 | 25 | 2729.812 | 1254.056 | 9.471 | 0.000 |

**Fig. 1**. Life cycle diagram used to project the Balearic shearwater population (pre-breeding census). Birds indicated age-stage classes: N1: individuals 1-year-old, N2: 2-years-old, $N3_{NB}$: 3-years-old not recruited, $N4_{NB}$: 4-years-old not recruited, $N5_{NB}$: 5-years-old not recruited, $N_B$: breeders, $N_{NB}$: animals in sabbatical that had bred at least once. ϒ: sabbatical probability, $r_{3,4,5}$: recruitment probability (probability of breeding for the first time) at 3-, 4- and 5-years-old, respectively; ρ: hatching sex ratio, S1: immature individual survival (first and second year of life); S2: adult survival for a non-breeder; S3: adult survival for a breeder; f: fecundity (fledging/female × year).

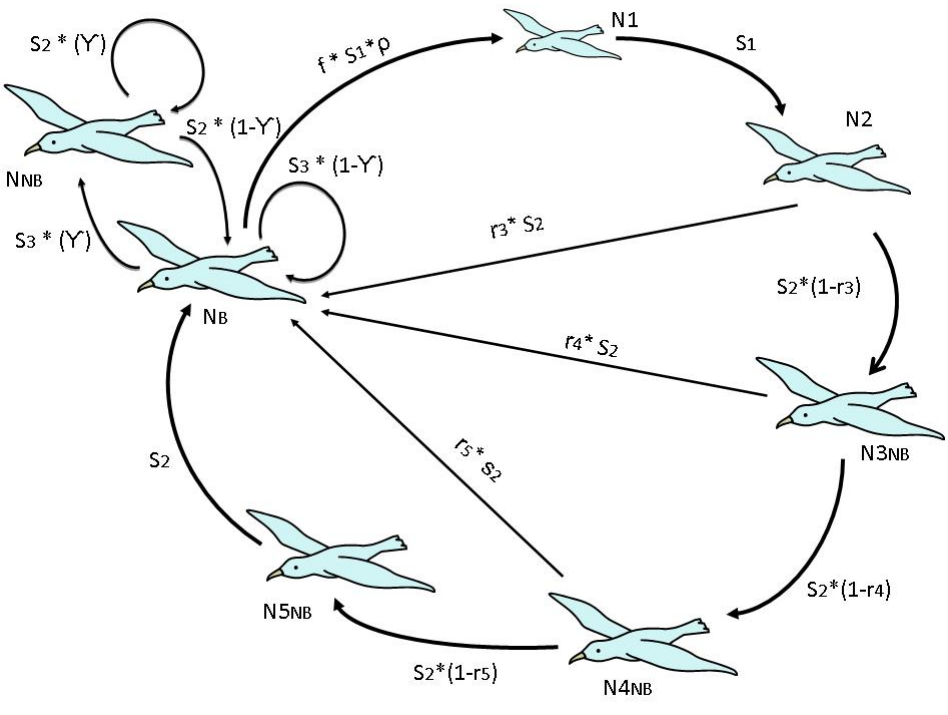
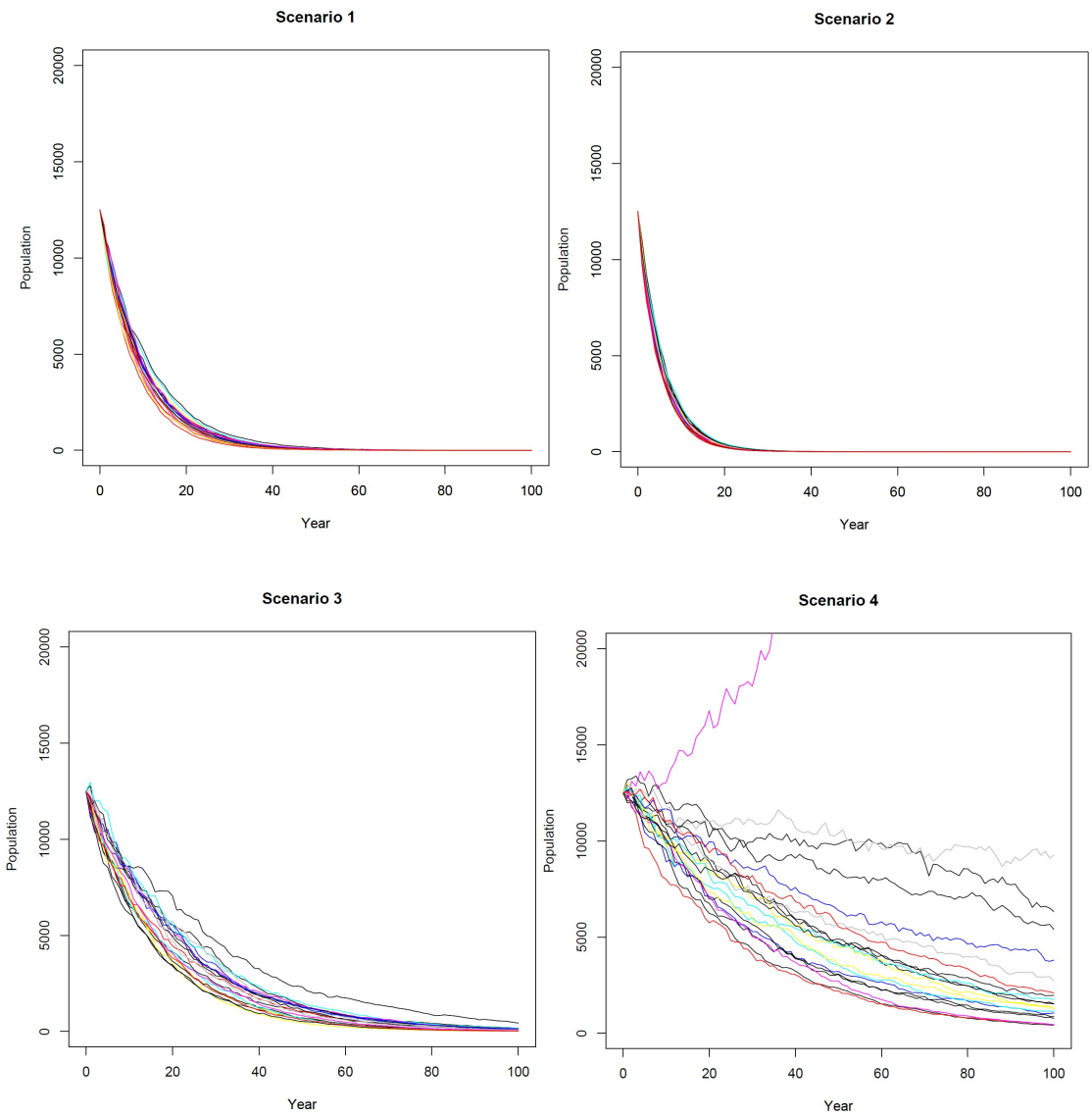
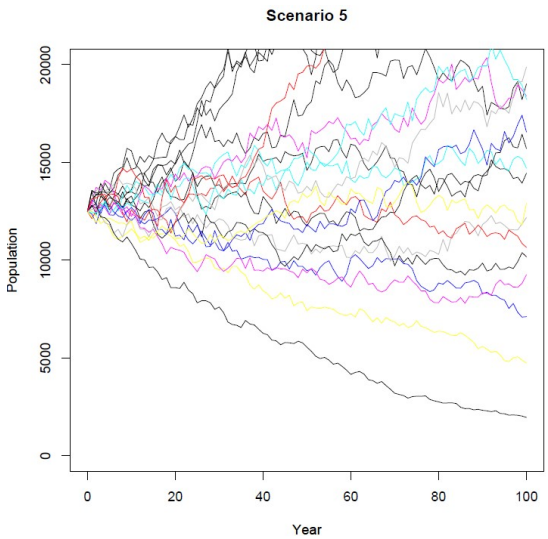
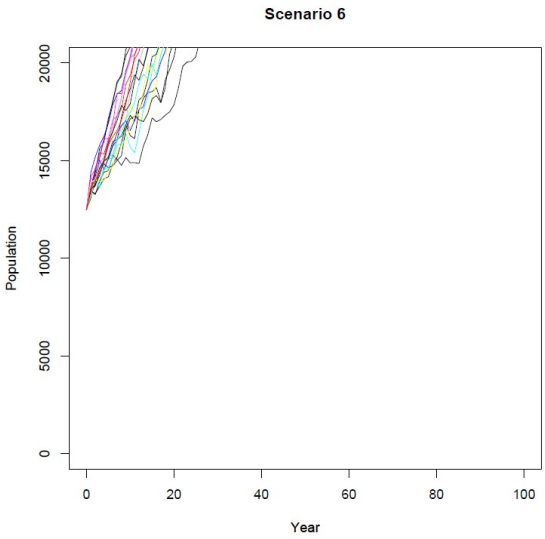


**Fig. 2.** Stochastic projections of Balearic shearwater population over 100 years under different scenarios proposed: 1) current situation; 2) reduced breeding success under future ban of discards; 3) conditions under future ban of discards but bycatch reduced; 4) current situation and bycatch reduced; 5) hypothetical conditions with optimal survival probabilities and discard banning; 6) current conditions but with optimal survival probabilities. Each graph shows 20 randomly chosen trajectories from the 1000 population trajectories run in our Monte Carlo simulations.




Scenario 5
Population
Year
0
5000
10000
15000
20000
0
20
40
60
80
100


Scenario 6
Population
Year
0
5000
10000
15000
20000
0
20
40
60
80
100

**Supporting Information**

Additional supporting information may be found in the online version of this article:

Appendix S1. Specification of the multi-event modelling approach in program E-SURGE

Table S1. European Union fisheries reports and plans that relate to the study.

Table S2. Population matrix model defining the average dynamics of the Balearic shearwater population.

Table S3. Survival estimates for other *Puffinus* shearwater species.

Table S4. Estimates of demographic parameters for the three best models in terms of AIC in the Recruitment analysis.

Table S5. Estimates of demographic parameters for the best model in terms of AIC in the Global analysis.

Table S6. Bycatch events of Balearic shearwaters gathered from the Literature.

Table S7. Sensitivity and elasticity of the population growth rate ($\lambda$).

Fig. S1. Relationship between Balearic shearwater breeding success and trawling landings.