Dramatic decline of the bearded reedling, *Panurus biarmicus*, in Spanish Mediterranean wetlands

R. Belenguer Barrionuevo, G. M. López–Iborra, J. I. Dies & J. Castany i Alvaro

Abstract

Dramatic decline of the bearded reedling, Panurus biarmicus, *in Spanish Mediterranean wetlands*.— The apparent stability of the bearded reedling in Spanish inland wetlands contrasts with its threatened status in Spanish coastal wetlands. The species has already disappeared from some coastal areas in Catalonia and its situation is critical in the region of Valencia. In 2013 we studied the breeding populations in three wetlands in Valencia using two methods: census by exhaustive search of individuals (territory mapping) and distance sampling using line transects. We estimated the trend of these populations from data obtained in previous censuses (2005 and 2006), and assessed their viability in the medium and short term using count–based population viability analysis (PVA). Results were alarming in the three studied wetlands, especially in the Albufera de Valencia, where only one breeding pair was found. The percentage of decrease of estimated pairs was similar in all wetlands: *ca*. 90% between 2005 and 2013. Results from the PVAs predicted a 90% probability of reaching the quasi–extinction threshold before 2024 or 2028 for the largest population of bearded reedling in the Valencia region, El Hondo, while for the Santa Pola population this threshold would be reached before 2016 or 2017. The parallel trend and generalized decline in the Spanish coastal wetlands suggests that these Mediterranean wetlands probably share some specific factors that have adversely affected their populations. Given that all these natural spaces are surrounded by intensively irrigated crops that are subjected to the intense use of pesticides, we hypothesize that these products could have had a detrimental effect on the bearded reedling. This hypothesis is supported by the fact that the healthiest populations are situated in Iberian inland wetlands that are surrounded by dry crops, where the use of pesticides is less intense. We propose cataloguing the species as Endangered at regional level.

Key words: *Panurus biarmicus*, Population decline, Censuses, Coastal wetlands

Resumen

Descenso acusado del bigotudo, Panurus biarmicus*, en el litoral mediterráneo español*.— La delicada situación del bigotudo en los humedales litorales españoles contrasta con la aparente estabilidad de la especie en los humedales interiores. La especie ha desaparecido de algunos enclaves costeros de Cataluña y su situación es crítica en la Comunidad Valenciana. En 2013 estimamos las poblaciones reproductoras en tres humedales de esta última comunidad usando dos métodos: un censo mediante búsqueda exhaustiva de individuos reproductores (mapeo de territorios) y un muestreo de distancia mediante transectos. Además, con los datos obtenidos en censos anteriores (2005 y 2006) estimamos la tendencia de estas poblaciones y calculamos su viabilidad a corto y medio plazo mediante un análisis de viabilidad poblacional (AVP) basado en conteos. Los resultados fueron muy preocupantes en los tres humedales estudiados, especialmente en La Albufera de Valencia, donde tan solo se encontró una pareja reproductora. El porcentaje de disminución de las parejas estimadas entre 2005 y 2013 en todos los humedales fue muy parecido, alrededor del 90%. Los resultados de los AVP para la mayor población de la Comunidad Valenciana, El Hondo, predicen un 90% de probabilidad de alcanzar el umbral de cuasi–extinción antes de 2024 o 2028, según el método de censo empleado. En el caso de Santa Pola, este umbral se alcanzaría antes de 2016 o 2017. El descenso generalizado y paralelo

en los humedales costeros españoles sugiere que probablemente estos hábitats comparten algunos factores específicos que han afectado negativamente a sus poblaciones de bigotudo. Puesto que todos estos espacios naturales están rodeados de cultivos intensivos de regadío sometidos a un elevado uso de plaguicidas, hipotetizamos que estos productos podrían haber tenido un efecto perjudicial en el bigotudo. Esta hipótesis se ve apoyada por el hecho de que las poblaciones más saludables están situadas en el interior peninsular, en humedales rodeados de cultivos de secano, en los que el uso de plaguicidas es menos intenso. Se propone recatalogar la especie a escala regional como En Peligro de Extinción.

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Introduction

Mediterranean reed–beds of common reeds, *Phragmites australis*, are particularly sensitive habitats with a high value for biodiversity conservation in Europe (Bibby & Lunn, 1982; Barbraud et al., 2002; Martínez–Vilalta et al., 2002; Poulin et al., 2002). They play an important role in the reproduction of many endangered passerine species, such as the moustached warbler, *Acrocephalus melanopogon*, the aquatic warbler, *Acrocephalus paludicola*, and the Eastern Iberian reed bunting, *Emberiza schoeniclus witherbyi* (Atienza & Copete, 2004; Castany & López, 2004; Tanneberger et al., 2010).

In recent years the extension and quality of this habitat in Europe has decreased for several reasons, mainly due to threats of human origin, such as pesticides, eutrophication, changes in land uses, alterations in water levels, and salinization (Poulin et al., 2002; López–Iborra & Monrós, 2004; Valkama et al., 2008). The reduction of reed–beds is one of the greatest threats for their dependent bird communities. The decrease and fragmentation of bird populations, along with the scarce exchange of individuals between these populations, could lead to loss of genetic diversity by endogamy, thereby reducing fitness (Johnson, 2001).

The bearded reedling, *Panurus biarmicus*, is a passerine occurring throughout Asia and Europe that lives in dense and well–preserved reed–beds (Robson, 2007). The European population is estimated at 240,000–480,000 breeding pairs, of which only around 1000 are found in Spain, inhabiting three main areas: Mediterranean coastal wetlands, inland wetlands in La Mancha, and northern small lakes (López–Iborra & Monrós, 2003; Birdlife International, 2012). In the Spanish Mediterranean coast, the bearded reedling appears irregularly in reed–beds from Catalonia to the south of Alicante, its southern limit of distribution in Europe. Until recently, it bred in the wetlands of l'Empordà, Utxesa, Ebro Delta, Albufera de Valencia, Salinas de Santa Pola and El Hondo (López–Iborra & Monrós, 2003). However, the main population in Catalonia (Ebro Delta) has reached extinction in the last decade, and populations reported for the wetlands in the Valencia region are small (López–Iborra et al., 2007). These Mediterranean populations seem to be virtually isolated because there is no evidence of movements of individuals between them (García–Peiró & López–Macià, 2002; López–Iborra & Monrós, 2004).

Given the concern regarding the situation of the species in Iberian Mediterranean wetlands it is necessary to periodically evaluate the size of populations in order to adopt conservation measures if needed. This task is complicated by the lack of a standardized methodology for use by diverse authors and repeated over time to obtain comparable results. Bearded reedling populations have been estimated by territory mapping in England (Gilbert et al., 1998; Ogilvie et al., 2004) and by line transects in Spain (López–Iborra et al., 2007). In the latter case, the mean density calculated for reed–beds or other potential habitats is extrapolated to the total area of these habitats in the studied wetlands. However, to our knowledge, the reliability of this approach has not been tested in wetlands. Thus, the main objectives of the present study were (1) to estimate the breeding populations in the study area using two methods: census by exhaustive search of individuals (territory mapping) and distance sampling using line transects, and (2) to assess the viability of the remaining Spanish populations nesting in coastal Mediterranean wetlands in the medium and short term.

Material and methods

Study area

This work was undertaken in the natural parks of Albufera de Valencia (39º 20' N, 0º 21' W), Salinas de Santa Pola (38º 11' N, 0º 38' W) and El Hondo (38º 11' N, 0º 45' W). All of three parks are protected wetlands on the Spanish Mediterranean coast (fig. 1). Albufera de Valencia is a shallow coastal lagoon situated south of Valencia city. Salinas de Santa Pola and El Hondo belong to an ancient lagoon south of the Alicante province. Santa Pola is taken up by artificial salt lakes built for sea salt production and game and fishing estates. El Hondo is occupied by two major regulating irrigation reservoirs and smaller ponds placed around the main ones.

The predominant vegetation is the common reed in all the wetlands, but southern cattail *Typha domingensis*, great fen–sedge *Cladium mariscus* and rushes *Juncus* spp*., Scirpus* spp. form some significant patches in Albufera de Valencia.

Population estimates

Transects were performed by walking close to the pond shores in Salinas de Santa Pola and El Hondo and by using a small boat in Albufera de Valencia. All potential habitats formed by reed–beds surrounding the major water bodies were sampled. Transects were aimed at covering the species' potential habitats during the breeding season (April to June) of 2005, 2006 and 2013. All transects were visited once in 2005 and 2006 (López–Iborra et al., 2006, 2007) and twice in 2013 (see location and length of transects in figure 1 and table 1).

All passerines were counted and their perpendicular distance to the transect line was estimated. The distribution of these distances was used to fit a detection function that estimates detection probability and species density, under a number of assumptions, using the software DISTANCE 6.0 (Buckland et al., 2001; Thomas et al., 2010). The detection curve was fitted using two of the key functions available in the software, half–normal and uniform, and cosine series expansion. Akaike Information Criterion (AIC) was used to select which of the two provided the best fit (Burnham & Anderson, 2002). We calculated 84% and 95% confidence intervals that robustly mimic 0.05 and 0.01 α values in statistical tests, respectively, for asymmetric confidence intervals (MacGregor–Fors & Payton, 2013). The reed–bed patches (including southern cattail and great fen–sedge when present)

were digitized from 1:5,000 orthophotos, resulting in 773 hectares in El Hondo, 212 in Santa Pola and 121 in Albufera de Valencia.

Territory mapping was carried out only in April and May of 2013. All the potential habitats were exhaustively searched in an attempt to locate the maximum number of bearded reedlings. In El Hondo and Albufera de Valencia, routes were made by boat, carefully prospecting the perimeter of the lagoons and large reed islands twice during the breeding season. A total distance of 73 km was covered in El Hondo and 21 km in Albufera. We also surveyed 29 points uniformly distributed through potential habitats in El Hondo, where the habitat is more closed and inaccessible than in the other studied wetlands, to discard the presence of birds that could have gone unnoticed during the censuses. Five–minute observations, with song and call broadcasting, were performed at these points, following the method described in Atienza (2006).

Population viability analysis

We evaluated quasi–extinction likelihoods of the bearded reedling at every wetland by extrapolating population trends from available census data (the so called count–based population viability analysis; Morris & Doak, 2002). For this purpose we used two methods. In the first, we estimated the instantaneous population growth rate (r) between two consecutive censuses using the equation

$$r = (ln(N_{t+a}) - ln(N_t))/a$$

where N_t is the population size in year t and a is the number of years between two consecutive censuses in the same wetland. We used the results of the population estimates of 2005 and 2006 (López–Iborra et al., 2006, 2007) and 2013, obtaining a total of six estimates of r (two per wetland). Population decline was simulated by multiplying the estimated number of pairs in each wetland by the value of λ

$$λ = e^r$$

where the value of r was randomly selected between the six estimates previously described. This process was iterated to simulate a 50–year period. Since the declining rate was very similar between wetlands (see results) we considered that these six estimates represent a sample of the existing variability in instantaneous population growth rate in the set of studied wetlands. We repeated this procedure 1,000 times and recorded the year when the population reached two individuals, considered as the quasi–extinction threshold.

The second method we applied was the diffusion model of Morris & Doak (2002), which estimates the extinction probability from the instantaneous population growth rate and its variance. These parameters were obtained from the series of abundance estimates (2005, 2006, and 2013) at each wetland. This model assumes that these parameters are constant over time, so that the resulting extinction probability corresponds to that expected if the population trend does not change, a condition shared with the previous model. Additionally, it is supposed that density–dependence does not exist,

which is reasonable in small declining populations that are well below the carrying capacity, as is the case here. To apply this model we used the popbio package (Stubben & Milligan, 2007) in R (R Development Core Team, 2009). We estimated instantaneous population growth rate and variance for each wetland applying the regression procedure described in Morris & Doak (2002) for variable time intervals between censuses. As in the previous method, we considered a quasi–extinction threshold of two individuals.

Both methods were applied three times in each locality, using the population size obtained in each study year as the initial population. The exception was Albufera, where population in 2013 had reached the quasi–extinction threshold (one pair). For simulations starting in 2005 and 2006 we used the population estimated from transects but for 2013 we used the results from exhaustive censuses.

Results

Population estimate

To estimate density, we first fitted a different detection probability to the original distance data at every locality (all years pooled) using the half–normal function. The Kolmogorov–Smirnov test for the q–q graphic was not–significant at any locality ($p > 0.4$), nor was the $χ^2$ test for any distance interval defined by the program ($p > 0.18$ in the worst case), so we worked with the original distances recorded in the field. The detection probabilities estimated by these models were very similar between localities (Albufera: 0.386; El Hondo: 0.383; Santa Pola: 0.383) and showed considerable overlap in their confidence intervals. We then fitted models with the same detection probability for all the study wetlands and tested the uniform and half–normal functions. The uniform function produced a lower AIC value, although the difference with the half–normal function was small (0.89), so the former was used to estimate density and population size for each year. The detection probability estimated by this model was 0.381 (confidence interval 95%: 0.336–0.437).

Densities were similar in the three wetlands. No statistically significant differences were found regarding density with the exception of Albufera in 2013, when no bearded reedlings were detected in the transects (table 2). Comparing years within wetlands, the 2005 and 2006 densities were not statistically different, according to the 84% confidence intervals ($p < 0.05$). However, the 84% confidence intervals for density in 2013 did not overlap with those of 2006, indicating a significant decrease between these years. The decrease in estimated pairs during these years was similar in the three wetlands: 88.9% in Albufera de Valencia (considering in this place the pairs estimated in 2013 by territory mapping), 86.6% in El Hondo, and 92.3% in Salinas de Santa Pola.

Population sizes estimated in 2013 using the territory mapping method were similar to those estimated from transects and well within their confidence intervals in El Hondo and Santa Pola (table 2). However,
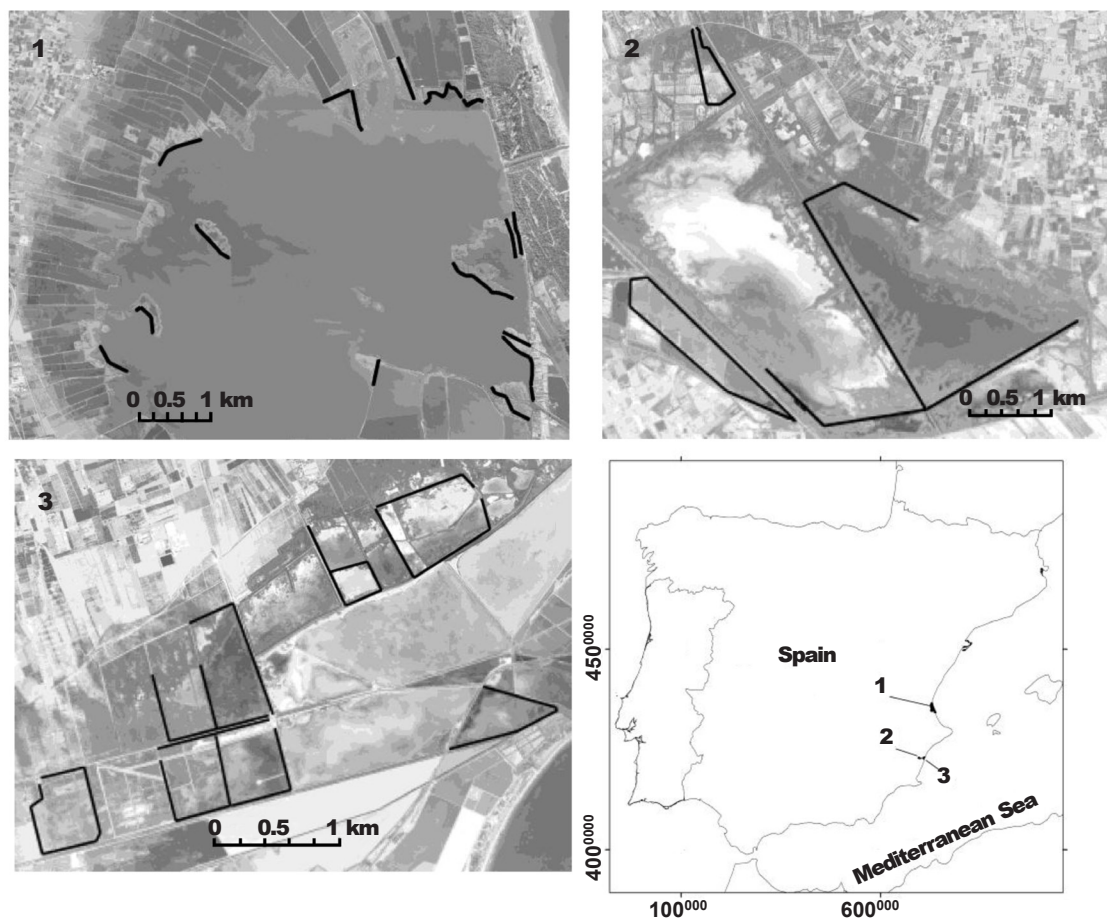
Fig. 1. Distribution of the transects in each wetland (black lines on aerial photographs) and location of the studied wetlands in SE Spain: 1. Albufera de Valencia; 2. El Hondo; 3. Salinas de Santa Pola.

Fig. 1. Distribución de los transectos en cada humedal (líneas negras en las fotografías aéreas) y localización de los humedales estudiados en el SE de España: 1. Albufera de Valencia; 2. El Hondo; 3. Salinas de Santa Pola.

territory mapping tended to give slightly higher values in all wetlands, including Albufera where only this method detected the presence of the species. The additional effort invested in El Hondo through the counting points with vocalization broadcasting did not produce any extra contacts.

Count–based population viability analysis (PVA)

The instantaneous growth rates estimated in each wetland by the Morris & Doak (2002) method were similar: Albufera $r = -0.2747$ (SE = 0.0542), El Hondo $r = -0.2515$ (SE = 0.0293), and Salinas de Santa Pola $r = -0.3206$ (SE = 0.0039). The PVA based on the diffusion model was done using the specific estimate of each locality. For the PVA based on simulations we used the set of the six estimates of instantaneous population growth rates estimated for the three wetlands. In this way we attempted to approximate to the

existing variability in growth rates. These estimates average -0.2822 (SE = 0.0532), a value placed within the range of rates estimated by the previous method.

Figure 2 shows the cumulative probability extinction function, *i.e.*, the probability that the population will have hit the quasi–extinction threshold at or before a given future time. The PVA results tended to be slightly more optimistic when population estimates of later years were used as initial values, although the predictions calculated using 2005 and 2006 values were very similar (table 3, fig. 2). The slightly later extinction times obtained with calculations starting in 2013 were due in part to the fact that we used the population estimated from the mapping method, which was somewhat larger than the results from transects. Comparison of the two models showed that the simulation procedure yielded more pessimistic results because the 0.5 and 0.9 thresholds of extinction probability were reached 3–5 years earlier than in the diffusion model, except in Santa

Table 1. Total length of transects and number of individuals detected in each location and year of study.

Tabla 1. Longitud total de los transectos y número de individuos detectados en cada localidad y año de estudio.

Zone	Locality	2005		2006		2013	
		Length	Individuals	Length	Individuals	Length	Individuals
Albufera	Norte	2.06	4	2.06	0	2.06	0
	Sur	1.35	0	1.35	1	1.35	0
	Este	4.02	2	3.52	2	3.52	0
	Oeste	1.25	3	1.25	2	1.25	0
	Total		9		5		0
El Hondo	Levante	4.10	4	4.10	1	4.05	0
	Poniente	2.10	0	2.10	0	2.10	0
	Charca SW	0.60	2	0.60	2	0.65	0
	Reserva	1.10	8	1.10	3	1.15	0
	La Raja	5.30	8	5.50	14	5.45	7
	Claudio					2.20	0
	Franja					0.50	2
	Total		22		20		9
Santa Pola	Santafé	4.22	5	4.49	19	5.45	0
	Flota Alta	2.00	6	1.47	0	1.45	2
	Múrtulas			2.96	2	2.65	1
	Charcol			1.51	2	1.50	0
	Bras del Port			1.56	0		
	Total		11		23		3

Pola where both models provided very similar results. Models estimate that the quasi–extinction threshold will be reached with 0.9 probability in a maximum of 11 years (simulation models) or 15 years after the last census (diffusion models).

Discussion

The two methods used in this survey (territory mapping and density estimation based on transects) produced consistent results across wetlands, although the transect method yielded slightly lower estimates in all cases. This, and the fact that in Albufera the species was detected only by territory mapping, suggests that this method was the most accurate. Territory mapping has been often used in other studies. In England, the national population of bearded reedling was estimated for the first time in 2002, using territory mapping and prospecting 71 localities (Gilbert et al., 1998; Olgivie et al., 2004). This methodology seems more feasible in small wetlands and with a high number of participants. However, in the case of large wetlands such as the Ebro Delta, Albufera de Valencia or El Hondo, transect methodology seems more practical and more economical because it can be carried out by a small team or a single person within a limited time given that it does not need to cover all the available habitat.

The results of line transects performed in the last years depict a population size reduction of 90% from 2005 to 2013 in all wetlands studied, indicating the critical situation of the species in this region. The results of the two different PVA methods that we used were very similar, and for the locality including the largest population of the bearded reedling in the Valencia region (El Hondo) they predicted a 90% probability of reaching the quasi–extinction threshold before 2024 or 2028, while for the Santa Pola population this threshold would be reached by 2017. The case of Albufera de Valencia is even more worrying because in 2005–2006 only between five and nine pairs were detected, and seven years later only one pair was found, indicating that this population is on the brink of extinction.

Results are based on only three estimates of breeding population sizes in each wetland, unevenly distributed along time, and thus they may have been influenced by the reduced population size estimated in the last study year, which could give a biased picture of the trend of the species. More censuses are therefore needed to have a better evaluation of the extinction

probabilities. However, the fragmentary information available for the years after our study supports our results. Although no more bearded reedling censuses have been carried out in Santa Pola and El Hondo, appropriate habitats of this last wetland are frequently surveyed by birdwatchers and researchers and this species continues to be very difficult to observe. In Albufera, the same transects used in this study were visited by JIDJ and JCA in 2014 and 2015 and they only detected one individual in 2014 and none in the last year. In this same wetland, independent birdwatchers and the staff of the Tancat de la Pipa Reserve have estimated 1–2 pairs in these two years (P. Vera, SEO/BirdLife, pers. comm.). Therefore, fortunately the species is still present in Albufera, but its numbers remain at or very close to the population size we estimated in 2013, which is the quasi–extinction threshold considered in our PVA models.

Given that the Albufera population had reached the quasi–extinction threshold by 2013, we used this result to test which of the two PVA models produced more realistic predictions. The simulation model gave a probability of 0.64 that the species would reach quasi–extinction threshold by 2013 starting with the 2005 population and 0.832 with the 2006 population, and the maximum probability of extinction was predicted to occur in 2014 and 2013 respectively. The equivalent values obtained from the diffusion model were 0.175 (2005) and 0.262 (2006) and the years of maximum probability of extinction were 2014 and 2015, respectively. These results suggest that the simulation model predicted the probability of extinction in Albufera more closely, although the differences between the two models are not very large due to uncertainties associated with this type of analysis.

A similar severe negative trend occurred before in another Spanish coastal wetland, the Ebro Delta, where in the eighties the species was considered a common breeder but is currently extinct (Martínez, 1983; Martínez & Elliot, 2004; Clarabuch, 2011; R. Gutiérrez, pers. comm.). This situation contrasts with the Iberian inland wetlands where the populations of this passerine are stable, as in the case of Castilla–La Mancha, and even experienced a recent expansion, as in northern Spain (Gutiérrez Expósito, 1998). Although the population trends in the French Mediterranean wetlands have not estimated as far as we know, the results of the census carried out in 2012 in southern France by the Tour du Valat research center, suggest that the bearded reedling is still relatively common there (B. Poulin, pers. comm.).

The existence of movements of individuals between wetlands would make a rescue effect possible and would be important for the viability of the Mediterranean populations. However, recoveries in Cataluña of birds ringed in southern France are scarce (only one occurrence connecting Aiguamolls de l'Empordà with the French Mediterranean wetlands), suggesting that movements between these wetlands have little relevance and that birds arriving from France are unlikely to have a rescue effect on the Catalonian populations (López–Iborra & Monrós, 2004). In north Spain, in contrast, despite the absence of recover-

Table 2. Densities (D, ind/ha) and standard errors (in parentheses) of bearded reedling estimated from the models generated by the program Distance. The population (numbers of pairs) estimated from the density values, P(D), with several confidence intervals (CI) is also shown, as well as the number of pairs estimated in 2013 by territory mapping (TM).

Tabla 2. Densidades (D, ind/ha) del bigotudo estimadas a partir de los modelos generados por el programa Distance, entre paréntesis se muestra el error estándar. Se muestran la población (número de parejas) estimada a partir de la densidad, P(D), con varios intervalos de confianza (CI) y las parejas estimadas en 2013 mediante el mapeo de territorios (TM).

	Year		
	2005	2006	2013
Albufera			
D	0.15 (0.07)	0.09 (0.04)	0.00
CI 95%	0.06–0.38	0.04–0.20	
CI 84%	0.08–0.28	0.05–0.16	
CI 76%	0.09–0.25	0.05–0.14	
P(D)	9	5.5	0
CI 95%	3.5–23	2.5–12	
CI 84%	4.5–17	3–9.5	
CI 76%	5–15.5	3–8.5	
TM			1
El Hondo			
D	0.22 (0.10)	0.20 (0.10)	0.03 (0.01)
CI 95%	0.08–0.64	0.06–0.64	0.01–0.09
CI 84%	0.11–0.45	0.09–0.43	0.01–0.06
CI 76%	0.13–0.39	0.11–0.38	0.02–0.05
P(D)	86	78.5	11.5
CI 95%	30–248.5	24.5–249	4–34
CI 84%	43–173	36.5–167.5	5.5–24
CI 76%	49–152	42–145.5	6.5–21
TM			14
Santa Pola			
D	0.18 (0.13)	0.14 (0.07)	0.01 (0.01)
CI 95%	0.05–0.70	0.05–0.37	0.00–0.05
CI 84%	0.07–0.46	0.07–0.28	0.01–0.03
C 76%	0.09–0.39	0.08–0.25	0.01–0.03
P(D)	19.5	15	1.5
CI 95%	5–74.5	6–39.5	0.5–5.5
CI 84%	7.5–49	7.5–29.5	0.5–3.5
CI 76%	9–41.5	8.5–26.5	0.5–3
TM			2

Table 3. Years in which the maximum probability of quasi–extinction and cumulative quasi–extinction probability thresholds of 0.5 and 0.9 were reached for each model and starting year.

Tabla 3. Años en los que se alcanzaron los umbrales de máxima probabilidad de cuasi–extinción y de probabilidad acumulada de cuasi–extinción, de 0,5 y 0,9 respectivamente, para cada modelo y año inicial.

	Diffusion model			Simulation model		
	2005	2006	2013	2005	2006	2013
Albufera						
Extinction *P* 0.5	2015–2016	2014–2015		2012–2013	2012–2013	
Extinction *P* 0.9	2019–2020	2018–2019		2014–2015	2013–2014	
Max. Extinction *P*	2015	2014		2014	2013	
El Hondo						
Extinction *P* 0.5	2022–2023	2023–2024	2023–2024	2019	2019–2020	2021–2022
Extinction *P* 0.9	2026–2027	2027–2028	2026–2027	2021–2022	2022–2023	2023–2024
Max. Extinction *P*	2023	2023	2024	2019	2020	2022
Santa Pola						
Extinction *P* 0.5	2014–2015	2014–2015	2015–2016	2015–2016	2015	2015–2016
Extinction *P* 0.9	2015–2016	2015–2016	2015–2016	2016–2017	2016–2017	2016–2017
Max. Extinction *P*	2015	2015	2016	2016	2015	2016

ies, it is thought that the species' expansion in this region has been due to the arrival of individuals from the Atlantic wetlands of France (Gutiérrez Expósito, 1998; López–Iborra & Monrós, 2004). In the case of La Mancha, recaptures connecting wetlands in this region are more frequent and it has been suggested that they behave as a metapopulation system (López–Iborra & Monrós, 2004), which could facilitate the stability of their populations. In contrast, there are no recoveries of ringed birds between coastal and inland wetlands. The movements between Spanish Mediterranean wetlands seem to be rare and there are only sporadic sights in some wetlands where the species does not breed (López–Iborra et al., 2007). This situation makes it very unlikely that the minimum abundance found simultaneously in the studied wetlands in the last study year is a consequence of temporal movements to other wetlands and increases the vulnerability of these populations to environmental changes.

Taken together, all these observations indicate that the Mediterranean bearded reedling populations in the Iberian peninsula have suffered a strong decline that contrasts with the steady situation, or even increasing trend in the other populations. These Mediterranean wetlands probably share some specific factors that have negatively affected other Iberian coastal populations of this species. All these wetlands are surrounded by intensively irrigated crops (rice, vegetables and fruit trees) where the use of pesticides is intense. The Ebro Delta and Albufera de Valencia are surrounded by rice fields extending to the very shore of the wetlands. In the Ebro Delta, concentrations of pesticides in waters

and soils after treatments in the surrounding rice fields are high (Mañosa et al., 2001) and are suspected of being cause of bivalve mortality episodes (Köck–Schulmeyer et al., 2011). In Albufera de Valencia, a study of soils within this natural park detected the presence of pesticides whose origin were the citrus groves and rice fields (Gamón et al., 2003). El Hondo reservoirs are fed with water that has irrigated intensive vegetable crops before it enters channels that carry it to reservoirs for re–cycling. The excess of this same water is transported to the Salinas de Santa Pola to flood the fishing and hunting reserves where a small population of bearded reedlings remains. The potential for pesticide concentration in these two wetlands is thus very high, although studies are needed to evaluate this more precisely. However, studies on the water quality at El Hondo showed a high degree of eutrophication (Colmenarejo et al., 2007) and occasional mortalities of birds and fish have been attributed to illegal discharges of pesticides into the channel net connecting these wetlands (Sehumed, 1997). On the contrary, the largest Spanish populations are in areas of dry farming (cereals and vineyards) with less intense agricultural activities (Urbano, 2008). Chemical products not only affect fertility and survival of wildlife, but also decrease the abundance of invertebrates consumed by insectivorous species (Brikle et al., 2000; Boatman et al., 2004; Morris et al., 2005; Taylor et al., 2006; Henderson et al., 2009). Given this contrast between the use of pesticides in the areas surrounding the La Mancha and in the Mediterranean coastal wetlands, it seems reasonable to consider that this factor could
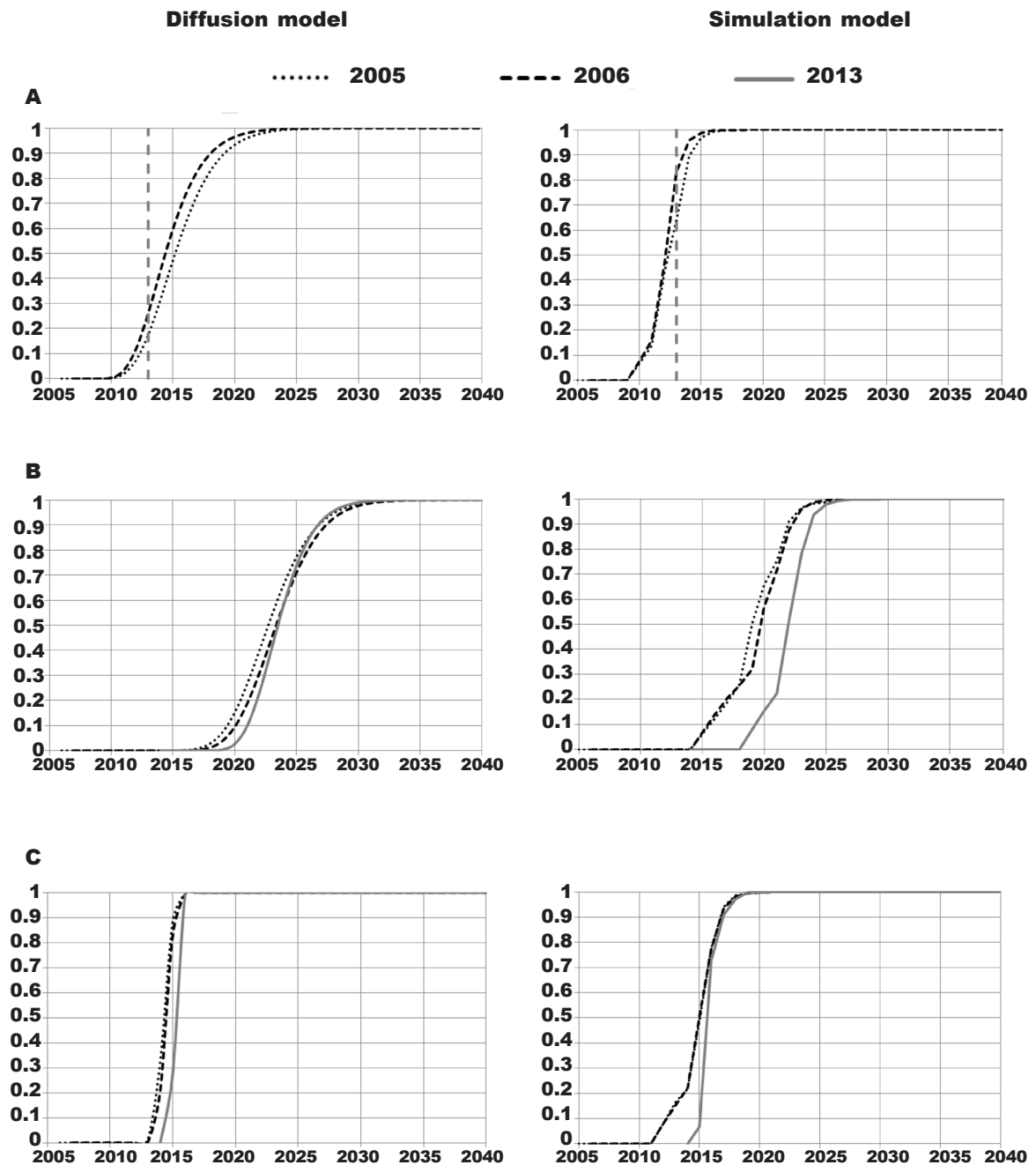
Diffusion model   **Simulation model**

······ **2005**   ---- **2006**   —— **2013**

A

B

C

Fig. 2. Cumulative quasi–extinction probability curves of the populations of bearded reedling in the three studied wetlands: A. Albufera; B. El Hondo; C. Santa Pola. Both PVA methods were applied using the population estimated in each census year as starting value (identified by patterns). In Albufera in 2013, the population had already reached the threshold of quasi–extinction (two individuals); thus PVA was not conducted for that year and the vertical dashed line (A) marks the year 2013 to facilitate the interpretation of the curves.

Fig. 2. Curvas acumulativas de probabilidad de cuasi–extinción de las poblaciones de bigotudo de los tres humedales estudiados: A. Albufera; B. El Hondo; C. Santa Pola. Ambos métodos de AVP se han aplicado usando la población estimada en cada año de estudio como valor inicial (identificados con tramas). Como en la Albufera de Valencia, la población ya había alcanzado el umbral de cuasi–extinción (dos bidividuos) en 2013; ese año no se realizó el AVP. La línea vertical discontinua (A) marca el año 2013 para facilitar la interpretación de las curvas.

have negatively affected coastal populations, because they are more exposed to pollutants than populations in inland wetlands. However, other causes of decline can not be discarded, such as increased predation by rats, which are effective predators of small passerine bird nests (Batáry et al., 2004; López–Iborra et al., 2004), and the degradation of reed–beds (Fernández–Núñez, 2014).

Given the reduced population size, the negative trend and the high extinction probability of bearded reedlings in Mediterranean coastal wetlands, it is urgent to implement effective measures to preserve the species. These should include appropriate management of the habitats to reduce nest losses during breeding. Water flow is mainly artificial in El Hondo and Santa Pola, and should thus be regulated to avoid strong water oscillations during the breeding season. Proliferation of opportunist predators, such as rodents, needs to be prevented and small reed islands for nesting should be promoted through reed management. However, for these measures to be effective the causes of the decline in these wetlands should be better understood. A decline of the magnitude observed in our study area corresponds to an 'Endangered' species status, according to the IUCN criteria (population size reduction of ≥ 50% over the last 10 years or three generations; IUCN, 2001). Therefore, we recommend that bearded reedling in Mediterranean regions (Catalonia and Valencia region) be upgraded from Vulnerable to Endangered. In addition, systematic and periodic censuses should be established in all the Spanish reproductive localities to detect future population regressions.

Acknowledgements

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