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Estimating feral cat density on Corvo Island, Azores, to assess the feasibility of feral cat eradication

Estimativa de densidades de gatos assilvestrados na Ilha do Corvo, Açores, para avaliar a viabilidade de erradicar a população de gatos assilvestrados

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ABSTRACT - Feral cats have had negative effects on native biodiversity on many islands worldwide. Eradicating feral cats from islands is often feasible, and can yield great benefits to native biodiversity, especially for seabirds. Corvo Island (Azores) is an important island where feral cats limit the distribution and abundance of breeding seabirds. To assess whether the eradication of feral cats on Corvo would be feasible we used camera traps to estimate the density of feral cats. We deployed 24 camera traps at 253 locations around the island for 14 months, and identified cats detected by camera traps individually based on the coat colour. We then used spatially explicit capture-recapture models to estimate cat density for Corvo. Cat density in the uninhabited upland part of Corvo, which is dominated by cow pastures, was 0.036 (95% CI 0.025 – 0.054) cats/ha. The lowland part of Corvo, which is inhabited by humans and contains domestic cats, had an estimated cat density of 0.734 (0.581 – 0.927) cats/ha. Overall, we estimated that the cat population on Corvo during our study period included 163 (123 - 228) individuals. The estimated cat densities are within the range of cat densities from other islands where cats have been successfully eradicated, and we conclude that feral cat eradication on Corvo would be technically feasible. However, the co-existence of feral and domestic cats would create operational challenges, and the current lack of a legal framework to ensure that all domestic cats are sterilised would increase the risk of a feral cat population becoming re-established after eradication.

RESUMO - Foram registados impactos negativos causados por gatos assilvestrados na biodiversidade nativa de várias ilhas do mundo. A erradicação destes de ilhas é muitas vezes viável, e pode trazer muitos benefícios para a biodiversidade nativa, especialmente aves marinhas. Na Ilha do Corvo (Açores) os gatos assilvestrados limitam a distribuição e abundância de aves marinhas nidificantes. Para determinar a sua densidade e a viabilidade de uma erradicação foram utilizadas câmaras-armadilha. Assim, foram colocadas 24 câmaras em 253 locais diferentes da ilha durante 14 meses. Os gatos detetados foram identificados individualmente pela cor do pelo. Através de modelos espaciais específicos de captura-recaptura foi estimada a densidade de gatos no Corvo. Na área de maior altitude, que é desabitada e dominada por pastagens com gado bovino, a densidade de gatos assilvestrados estimada foi de 0,036 (95% IC 0,025 – 0,054) gatos/ha. Na área de baixa altitude, que é habitada e contém gatos domésticos, a densidade de gatos assilvestrados foi de 0,734 (0,581 – 0,927) gatos/ha.

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No total, durante o período de estudo a população de gatos no Corvo foi estimada em 163 (123 - 228) indivíduos. A estimativa de gatos para o Corvo está dentro dos valores estimados para outras ilhas onde os gatos assilvestrados foram erradicados com sucesso, pelo que concluímos que a erradicação de gatos assilvestrados no Corvo é tecnicamente viável. No entanto, a co-existência de gatos domésticos e assilvestrados poderá ser um problema na logística operacional, além da falta de enquadramento legal para garantir a esterilização de todos os gatos domésticos, o que aumentará o risco de restabelecimento dos gatos assilvestrados após uma erradicação.

Feral cats (*Felis catus*) are a widespread introduced predator on many islands, and are known to have negatively affected many populations of native vertebrates (Bonnaud et al. 2010a, Medina et al. 2011). Because many vertebrates that are native to oceanic islands evolved in the absence of mammalian predators such as cats, they are extremely vulnerable to cat predation and many cannot co-exist with feral cats. Seabirds in particular have suffered greatly from cat predation (Keitt et al. 2002, Bonnaud et al. 2009), and the removal of feral cats from islands is often the most efficient way to preserve populations of vulnerable seabirds (Keitt & Tershy 2003, Bonnaud et al. 2010b, Ratcliffe et al. 2010).

Minimising or eliminating the negative effects of feral cats on native taxa on oceanic islands requires knowledge about the biology of feral cats in order to target control or capture techniques effectively (Nogales et al. 2004, Campbell et al. 2010). One important aspect of cat biology that aids in designing effective control or removal programmes is the ranging behaviour and the density of feral cats, as this will determine the efficient placement of traps or poison bait stations to ensure every single cat has access to such a control device (Moseby et al. 2009, Bengsen et al. 2012b). However, estimating the density of a solitary and far-ranging carnivore like feral cats is logistically challenging (Obbard et al. 2010, Can et al. 2011, Sollmann et al. 2011).

Corvo Island (39°42'N 31°6'W; 1760 ha) is the smallest island in the Azores, and is home to a large population of seabirds (Monteiro et al. 1996, Groz et al. 2005). Feral cats were introduced to Corvo before 1717, and are continuously supplanted by offspring from an un-neutered domestic cat population. Feral cats are currently the main predators of the most abundant seabird on Corvo, the Cory's Shearwater (*Calonectris diomedea*; Hervías et al. 2013), and are presumably the main reason why other species of smaller seabirds are restricted to cliffs that are inaccessible to feral cats (Groz et al. 2005). The eradication of feral cats from Corvo Island may therefore benefit the local seabird community.

In this study, we estimated the density of feral cats on the island of Corvo in order to assess whether a feral cat eradication would be technically feasible. We used camera traps for 14 months to record cats at various points across the island, and identified individual cats based on coat colour pattern. These spatially referenced photograph data were then used to estimate cat density with spatially explicit capture-recapture models. Our study provides a first estimate of the number of feral cats on Corvo, and describes the challenges associated with cat density estimation that may be useful for similar assessments elsewhere.

METHODS

» Study area

Corvo is geographically dominated by the dormant volcano that originally created the island (França 2006). The most dominant habitat is grassland grazed by cows, followed by small agricultural plots and cliffs vegetated by native small shrub or invasive vegetation such as Giant Reed Grass (*Arundo donax*). There are no forests on Corvo, and only small (<1 ha) woodlands along riparian areas. Corvo ranges in elevation from sea level to 718 m, and the only village (437 inhabitants) and most agricultural areas are situated in the lower part of the island (<150,

Fig. 1). Because the populated and more intensively farmed area of Corvo has habitat characteristics that are distinctly different from the remainder of the island, we separated the island into a lowland (below 150 m) and an upland part (above 150 m), and estimated cat density for these two parts separately (Fig. 1).

» Measuring cat activity using trail cameras

We used passive infra-red motion sensor cameras (Bushnell Trophy Cam) triggered by temperature alterations and movements to detect feral cats. We set the delay between each trigger to ten seconds and recorded three images per trigger event to maximise identification of cats and to reduce excess images from slow-moving animals. Cameras were placed 50 cm above ground and attached to fixed objects (rocks, fence-posts, trees), and no attractants were used to lure cats in front of cameras. To achieve greater spatial coverage, cameras were moved to a new position every 2 weeks.

From May through October 2010, cameras were deployed at six shearwater colonies to assess predator activity (Hervías et al. 2013). However, to estimate cat density across the entire island we also deployed cameras at random locations around the island outside the breeding season (November 2010 – April 2011). Spatial deployment of cameras during the breeding season was based on a 15 x 15 m grid that was created for each shearwater colony. For every 2-week period, each camera was assigned to a randomly chosen grid cell, with 2-4 cameras operating simultaneously in a given colony. Within the randomly chosen grid cell, cameras were placed along paths or boundaries to maximise the likelihood that a lateral view of a passing cat could be obtained. Outside the breeding season, cameras were placed at randomly generated locations around the entire island (Fig. 1).

The images downloaded from cameras were processed manually to filter out images that contained cats (<1% of all images), and each cat was individually identified by its unique coat markings (Mendoza et al. 2011). To match new images with previously identified individuals we manually compared each new image with all others in an existing archive of cat images. Animals that could not be matched to an existing individual in the archive were given a new identification number. Some cat detections involved cats that were not uniquely identifiable, due to uniform coat colour that did not exhibit distinguishable characteristics. These detections received a unique identification number indicating the observation of an unidentifiable cat, and were excluded from the analysis of cat density. To account for the number of unidentifiable cats in the population, we calculated the proportion of cat detections that involved indistinguishable cats. The final estimates of cat density were then divided by the proportion of unidentifiable individuals in our sampled population, assuming that the total feral cat population would harbour the same proportion of unidentifiable cats as the sampled population (Wilson et al. 1999, Gormley et al. 2005).

» Cat density estimation using spatially explicit capture-recapture models

Spatially explicit capture-recapture (SECR) models use the individual identification of animals at different locations within the study area to estimate the approximate location of each individual's home range centre (Efford et al. 2009, Royle et al. 2009, Borchers 2012). These methods utilise additional spatial information compared to standard capture-recapture analyses to estimate abundance, and require no *a priori* definition of the effective sampling area (Borchers 2012).

We deployed cameras at 125 locations in the upland part and at 128 locations in the lowland part. Cameras were deployed for at least 14 nights at each location, and we reduced continuously acquired camera images into encounter occasions spanning 7 days, yielding 57 discrete encounter occasions between March 2010 and April 2011. The length of these encounter occasions was chosen to maximise the number of detections per encounter occasion, while minimising the loss of information from cats being repeatedly recorded during a single occasion (Bengsen et al. 2012a). The long study duration (> 1 year) was required to obtain sufficient recaptures of previously identified animals for the estimation of density.

One of the major assumptions of SECR models is that the population is closed during the study period. Although Corvo is an island and emigration and immigration of feral cats is highly unlikely, the

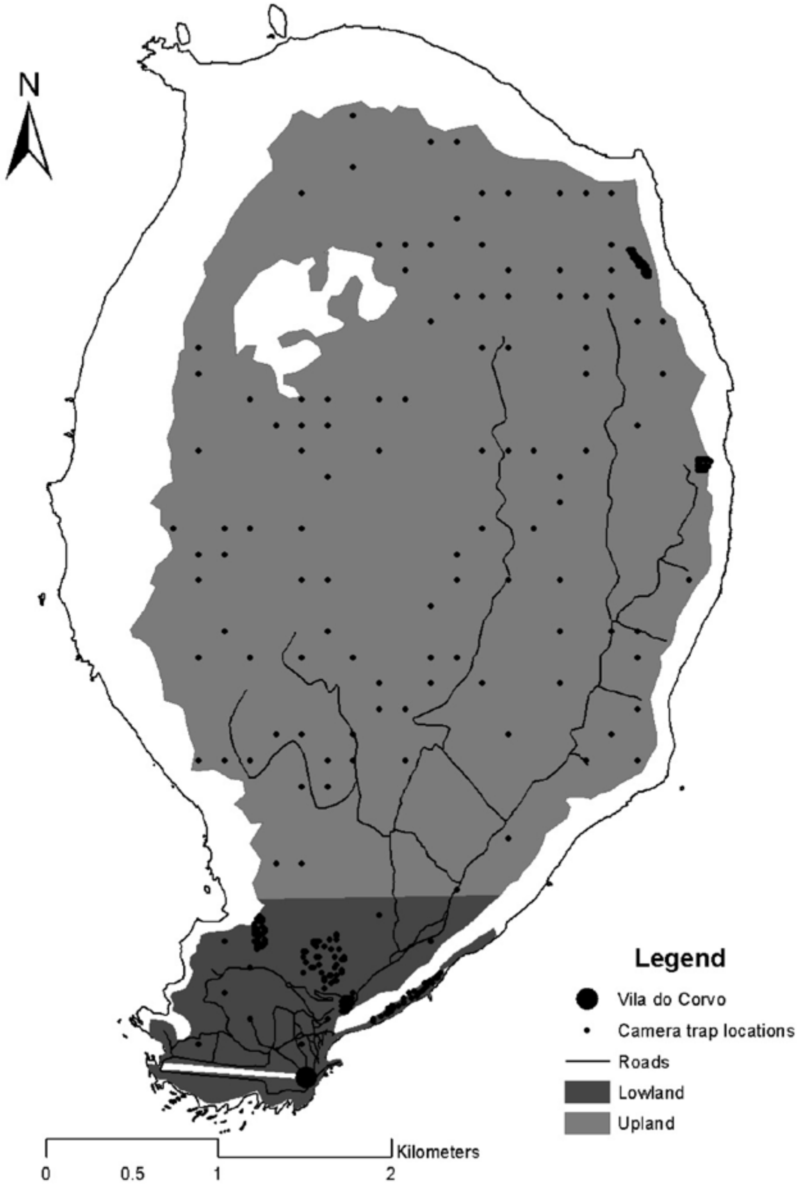

Figure 1. Map of Corvo showing the location of the village, the roads, and all camera trap locations used during this project. The area in dark grey represents the lowland habitat, and the area in light grey the upland habitat for which cat density was estimated. White areas were excluded from cat density estimation because they were inaccessible to humans or cats (e.g. vertical cliffs, the crater lake, and the airport). Dense aggregation of camera traps occurred in study colonies of Cory's Shearwaters.

Figura 1. *Mapa do Corvo com a localização da vila, as estradas e os locais de colocação das câmaras. A densidade de gatos foi estimada no habitat de baixa altitude, área representada em cinzento escuro, e a área em cinzento claro representa o habitat de alta altitude. As áreas em branco foram excluídas do estudo por serem inacessíveis a humanos e gatos (p.e. falésias, o Caldeirão e o aeroporto). As áreas com uma maior densidade de câmaras correspondem às colónias de cagarra.*

long study duration rendered it likely that the feral cat population was not demographically closed, i.e. that births and deaths occurred during the study period. We used a closure test to evaluate whether our data met the assumption of population closure (Stanley & Burnham 1999). If the closure assumption was violated, we adopted the reduced version of SECR model that collapses all occasions into one and simply uses the spatial encounter history of individuals (Borchers & Efford 2008).

We evaluated two different spatial detection functions (half-normal and exponential), and two different functions for the distribution of home range centres, a Poisson point process (Borchers & Efford 2008) and a binomial point process (Royle et al. 2009). We thus fit four different candidate models combining each combination of detection function and distribution process. For numerical integration, the likelihood function was evaluated at points spaced 50 m apart and distributed evenly throughout the accessible parts of Corvo. We created a habitat mask that excluded areas from integration that are inaccessible to cats (open water in the crater lake, vertical cliffs, Fig. 1). All SECR models were fit in R 2.13.0 (R Development Core Team 2010) using the library 'secr' (Efford 2011).

For each part of the island we ranked models using AIC_c and estimated density by model-averaging across all candidate models using the Akaike weight of each model as a weighing factor (Burnham & Anderson 2002). For each part we estimated the number of cats by multiplying the estimated density by the area (1140 ha for the upland part, 139 ha for the lowland part), and correcting for the proportion of unidentifiable cats as described above. We present cat density and abundance estimates as model-averaged mean with 95% confidence intervals.

RESULTS

In the upland part of Corvo, we identified 30 individual cats in 83 detection events at 58 camera locations. An additional 21 detections involved unidentifiable individuals, yielding a proportion of 20.2% of all detections (n = 104) being not identifiable. In the lowland part of Corvo, we identified 74 individual cats in 352 detection events at 92 camera locations. An additional 43 detections involved unidentifiable individuals, yielding a proportion of 10.8% of all detections (n = 395) being not identifiable. The closure test indicated that the population in each part was not closed over the study period (upland: $z = -3.20$, lowland: $z = -1.17$, both $p < 0.001$), hence we reduced the encounter history of all detectors to a single occasion.

For each part of the island, the model that assumed a negative exponential detection function and a Poisson point process received the most support from the data (upland: $\omega AIC_c = 0.75$, lowland: $\omega AIC_c = 1.0$). The model-averaged density of feral cats was 0.036 (95% CI 0.025 – 0.054) cats/ha in upland Corvo, and 0.734 (0.581 – 0.927) cats/ha in the lowland part. Accounting for unidentifiable individuals we estimated that the feral cat population in the upland part contained 52 (35 – 77) individuals, while the lowland part contained 111 (88 – 141) individuals. Thus, the cat population on Corvo during our study period was 163 (123 – 228) individuals. This figure includes an unknown proportion of semi-domestic cats that venture out from the village and were included in our density calculation because they were detected on cameras. However, the figure does not include the truly domestic cats that remain in and around the houses of their owners and were never detected on cameras.

DISCUSSION

Our study provides the first estimate of the number of feral cats on Corvo. We estimated that 123 – 228 cats existed on Corvo during the study period, with a marked difference in density between the lowland and the upland part of the island. The lowland part had a cat density ~20 times higher than the upland part, which is unsurprising given the presence of the village and a large number of available food sources (rubbish dump, agricultural fields and livestock pens that sustain high numbers of rats and mice) and places that provide cats with shelter from inclement weather. While the estimate of density in the upland part of the island probably reflects only feral cats, the lowland part is inhabited by both feral and domestic cats. The human population in the village of Corvo sustains a domestic (pet) and semi-domestic (regularly fed) cat population that together encompass 100 – 120 individuals (113 pet cats were officially registered between February and August 2010). Some of these cats roam around parts of the island, and were detected at camera locations in our study. Hence, our abundance estimate for the lowland part surrounding the village includes some of the owned cats on Corvo, which are not truly feral but fed by humans.

The cat densities that we estimated fall within the range of other islands where cats have been eradicated successfully (Nogales et al. 2004, Campbell et al. 2010). Particularly in the upland area, cat density appeared to be lower than on some sub-Antarctic islands where cats have been successfully eradicated (e.g. Macquarie Island: 0.2 cats/ha, (Copson & Whinam 2001); Marion Island: 0.1 cats/ha, (Van Aarde 1979), Kerguelen: 0.002 – 0.2 cats/ha (Say et al. 2002)). Although the cat density in the lowland part of the island was substantially higher, the estimated density is still much lower than densities of domestic cats estimated in urban areas (1.3 – 15.8 cats/ha; Sims et al. 2008). Thus, we believe that the eradication of feral cats from the island of Corvo would be technically feasible with currently available technology.

However, when considering the eradication of feral cats from an island, the presence of domestic and semi-domestic cats can create substantial operational obstacles. In addition, many of the cat owners on Corvo may oppose the eradication of feral cats, or it may not be practical to ensure that domestic cats will not re-establish a feral population (Oppel et al. 2011). Although the distinction between domestic and truly feral cats could be misleading from a conservation perspective, because domestic cats also prey heavily on local wildlife (Woods et al. 2003, van Heezik et al. 2010, Horn et al. 2011), the control and minimisation of cat predation on native fauna by feral and domestic cats requires different political and practical approaches (Calver et al. 2011).

The largest inhabited island on which feral cat eradication has been accomplished to date is Ascension Island (9700 ha, 859 m elev.), a U.K. overseas territory in the Atlantic Ocean with ~1100 human inhabitants (Bell & Boyle 2004, Ratcliffe et al. 2010). Although Corvo has only 437 human inhabitants, the problems experienced with domestic cats during the feral cat eradication campaign on Ascension are directly transferrable to the situation on Corvo. In particular, the use of poison bait to kill feral cats would be inacceptable due to the substantial risk of accidentally poisoning domestic cats (Ratcliffe et al. 2010). Hence, a feral cat eradication campaign on Corvo would have to rely on live trapping, which is generally more expensive and less efficient (Nogales et al. 2004, Campbell et al. 2010, Ratcliffe et al. 2010). Despite the inadvertent side-effects of the feral cat eradication, the Ascension Island operation may offer a viable and instructive example for Corvo with respect to domestic cats. On Ascension, all domestic cats are legally required to be registered, micro chipped and neutered, thus eliminating the risk of unwanted kittens being released into the wild and re-establishing a feral cat population (Ratcliffe et al. 2010). A similar effort to provide free sterilisation and micro-chipping of domestic cats on Corvo during 2010 received broad acceptance by most cat-owners on the island (51% of 113 domestic cats were sterilised), but found no support by the regional government. Hence, there is currently no legal framework and no available opportunity to sterilise the domestic cat population, which would render an eradication of the feral cat population an almost futile exercise with only temporary benefits for native wildlife.

» Methodological challenges in estimating cat density

Our approach to estimate cat density using spatially explicit capture-recapture models yielded realistic and moderately precise estimates of the number of cats on the island of Corvo. Such assessments are vital to allocate resources appropriately during control or eradication efforts (Moseby et al. 2009, Bengsen et al. 2012b). We encountered several challenges during the camera trapping, which may be instructive for similar projects elsewhere. The cost of camera traps (~US$200 per unit) restricted the number of cameras that could be employed in this project to 24, thus limiting the spatial coverage that could be achieved. We rotated cameras around locations to overcome that limitation, which led to the problem that at many locations no cats were detected because the camera was only present for 2 weeks. An alternative approach to maximise the detection probability of cats at camera traps would be to bait camera traps and entice cats to be detected (Bengsen et al. 2012a). However, baiting camera traps violates a core assumption of spatially explicit capture-recapture models (Efford et al. 2009, Foster & Harmsen 2012) and of other procedures to estimate density (Rowcliffe et al. 2008). These methods both model the natural movement process of cats, and thus require that recorded cat movements are unbiased and not distorted by a cat being attracted to a baited recording station. We therefore chose to deploy unbaited camera traps.

Ideally, mark-recapture studies are conducted over a time frame during which the population under study is closed. Possibly as a result of unbaited camera traps, the encounter rates of cats at cameras were too low to estimate density within a short time frame, and we therefore accumulated data over more than one year. Although it would have been possible to estimate density with a subset of our data, these estimates were generally too low because the number of individual cats identified during the entire study period was larger than the number estimated had we used only a subset of data.

Further challenges in our approach included a large number of images that did not contain any cats and required substantial effort to be filtered out. Excessive images resulting from cows or moving vegetation occasionally led to overload of available memory (2 GB), resulting in a truncation of the effective operation time because no more images could be stored. While some of these excessive images could be reduced by altering camera sensor settings, passing sheep or cows will always trigger cameras designed to detect small mammals. We received help from a large group of volunteers to screen thousands of images for the presence of cats, and caution practitioners interested in using this approach to not underestimate the amount of effort that is required to manually filter images. Besides detecting cats on images, the individual identification of cats presented another challenge. Although novel techniques continue to be developed to enhance the reliability of individual identification (Mendoza et al. 2011, Morrison et al. 2011, Goswami et al. 2012), we found that automated identification algorithms are difficult to apply to images of cats, because the angle, lighting, distance, and posture of cats differed between images, requiring a human to positively match individuals among different images.

Despite these challenges we believe that the resulting density estimates provide extremely useful information to advance the planning of cat control or eradication actions for the benefit of seabirds on the island of Corvo.

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Invasive mammal species on Corvo Island: is their eradication technically feasible?

Mamíferos exóticos na Ilha do Corvo: é a sua erradicação tecnicamente possível?

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ABSTRACT - The successful eradication of exotic mammals from an island depends on many factors: the number of invasive species, the number of individuals of each species, biogeography and size of the island, and the support of the government authorities and people inhabiting the island. The LIFE project "Safe Island for Seabirds" (2009-2012) attempted to investigate the feasibility of performing an eradication programme of the exotic mammals on Corvo Island. This study presents results of monitoring the black rat *Rattus rattus*, house mouse *Mus domesticus*, cat *Felis catus*, goat *Capra aegagrus hircus*, and sheep *Ovis aries*, carried out during the project. The relative abundance and distribution of rodents, goats and sheep was determined each month, and a sterilisation campaign was developed for cats. Our results showed that the eradication of rodents is technically feasible, and should be attempted in April (when the lowest peak in abundance occurred for both species). For cats, although a higher numbers of feral cats than those estimated for Corvo have been eradicated on other islands, there are several constraints that will have to be overcome - in particular, the limits of legal responsibility for the management of cats. The eradication of goats and sheep could be attempted, but due to the island's steep cliffs it would be very expensive. Education programmes must be carried out to inform the human inhabitants about the problems caused by exotic mammals for endangered species, since currently local government authorities do not see the eradication of invasive species as a political priority.

RESUMO - O sucesso da erradicação de mamíferos exóticos de uma ilha depende de vários fatores: número de espécies invasoras, número de indivíduos de cada espécie, biogeografia e tamanho da ilha, e o apoio das entidades governamentais e dos habitantes. O projeto LIFE "Ilhas Santuário para as Aves Marinhas" (2009-2012) procurou investigar a viabilidade de realizar um programa de erradicação de mamíferos exóticos na Ilha do Corvo. Este estudo apresenta resultados da monitorização do rato-preto *Rattus rattus*, do rato-caseiro *Mus domesticus*, do gato *Felis catus*, da cabra *Capra aegagrus hircus* e da ovelha *Ovis aries* levada a cabo durante o projeto. A abundância relativa e a distribuição de roedores, cabras e ovelhas foram determinadas mensalmente e foi efetuada uma campanha de esterilização dos gatos. Os nossos resultados mostraram que a erradicação de roedores é tecnicamente possível e deveria ocorrer em abril, no pico de menor abundância para as duas espécies. No caso dos gatos, apesar de a

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erradicação dos assilvestrados já ter sido feita noutras ilhas com números superiores aos estimados para o Corvo, existem vários obstáculos que terão que ser ultrapassados, nomeadamente a ausência de responsabilidade legal para a gestão dos gatos. A erradicação de cabras e ovelhas poderia ser conseguida, mas devido às íngremes falésias costeiras seria muito dispendiosa. Programas de educação deveriam ser implementados para informar a população sobre os problemas causados pelos mamíferos exóticos sobre espécies em perigo, uma vez que as atuais autoridades do governo local não consideram a erradicação de espécies invasoras como uma prioridade política.

Exotic mammals, as ecological generalists, have successfully colonized a wide range of habitats on islands and have caused irreversible damage to island biota (Atkinson 1985; Courchamp et al. 2003; Towns et al. 2006; Jones et al. 2008). The impacts of exotic mammals on native ecosystems vary among islands (Clark 1981; Fitzgerald 1988; Yabe et al. 2010) since they are influenced by the presence of other introduced species (Cuthbert 2002; Bonnaud et al. 2010). Eradication is often the preferred strategy for the removal of exotic species on islands (Brooke et al. 2007; Aguirre-Muñoz et al. 2009; Capizzi et al. 2010; Oppel et al. 2011; Veitch et al. 2011). But whether eradication is possible depends on many factors, including the support of the people inhabiting the island, which can be challenging.

The Archipelago of Azores (Portugal) provides nesting habitat for a total of eleven seabird species and some of them are now restricted to a few islets and remote coastal strips (Monteiro et al. 1996). Two of the most important threats for the current seabird populations breeding in the Azores are reduction of suitable breeding habitat due to human activities and predation by introduced mammals (Monteiro et al. 1996), which are probably causing ongoing population declines (Fontaine et al. 2011). Corvo, being the smallest island of the archipelago, it exhibits a well preserved coastal environment, contains a small population (437 inhabitants), and offers the best potential and conditions for seabird recolonization. Corvo still has important seabird colonies, including of many species that are classified as priority by the Annex I of the Birds Directive and as vulnerable by the World Conservation Union (Table 1). With only one village, Corvo has the biggest number and density of Cory's shearwaters *Calonectris diomedea* breeding in the Azores, but also smaller numbers of Little shearwater *Puffinus assimilis*, Manx shearwater *Puffinus puffinus*, Madeiran storm-petrel *Oceanodroma castro*, Roseate tern *Sterna dougallii* and Common tern *Sterna hirundo*. Besides these seabirds, there is also Azorean wood pigeons *Columba palumbus azorica* breeding on the island. As a result, Corvo has the highest percentage of appropriate areas to preserve Azorean avifauna (Rodriguez and Cunha 2012), is biosphere reserve since 2007, and the Nature Park was created the in 2008 under Regional Legislative Decree 44/2008/A, in order to conserve and protect species habitat and natural resources. It also has an Important Bird Area (IBA) which includes the coast in most of the island, two Special Protected Areas (SPAs) and one Site of Community Interest (SIC).

Among all seabird species breeding on Corvo Island, only Cory's shearwater breeds in locations that are accessible to humans and thus amenable to scientific investigations. A recent study about the impacts of exotic mammals on breeding success confirmed that predation by black rats *Rattus rattus* and feral cats *Felis catus* is the main cause of nest failure (Hervías et al. 2012). Predation by exotic mammals is thus a major concern, because adults of larger species, such as Cory's shearwater and Yellow-legged gull *Larus michahellis*, are probably less vulnerable to these predators than all other seabird species for which nidification has been confirmed on Corvo, which are all significantly smaller. As adult survival is likely to have a much stronger influence on population growth rate than nest success (Fontaine et al. 2011), these species are much more vulnerable to both rodent and

Table 1. Breeding seabird species in the nine islands of the Archipelago of Azores and their IUCN Red List Category; LC = Least Concern; VU = Vulnerable; NT = Near Threatened (IUCN 2012).
Tabela 1. *Aves marinhas nidificantes nas nove ilhas do arquipélago dos Açores e seu estatuto de ameaça de acordo com a Lista Vermelha da UICN LC = Pouco preocupante; VU = Vulnerável; NT = Quase ameaçado (IUCN 2012).*

Seabird species	Corvo	Flores	Faial	Pico	São Jorge	Graciosa	Terceira	São Miguel	Santa Maria	IUCN
Bulweria bulwerii						X			X	LC
Calonectris diomedea	X	X	X	X	X	X	X	X	X	LC
Larus michahellis	X	X	X	X	X	X	X	X	X	LC
Oceanodroma castro					X	X			X	LC
Oceanodroma monteiroi						X				VU
Onychoprion fuscatus						X			X	LC
Pterodroma deserta						X				NT
Puffinus assimilis	X	X	X	X	X	X		X	X	LC
Puffinus puffinus	X	X								LC
Sterna dougallii	X	X	X	X	X	X	X	X	X	LC
Sterna hirundo	X	X	X	X	X	X	X	X	X	LC

cat predation than Cory's shearwaters. Breeding colonies of Manx shearwater are restricted to Corvo and Flores islands (Monteiro et al. 1999). Monteiro's storm petrel *Oceanodroma monteiroi*, so far only known from Graciosa, may also breed here (hot season Azores population; Bolton et al. 2008) but evidence of breeding is still lacking.

Given the high conservation interest of seabird species on Corvo, an ambitious European Commission funded LIFE Project (07 NAT/P/000649) entitled "Safe Islands for Seabirds" was carried out from January 2009 to December 2012. This project aimed to create a management plan for the exotic mammals on Corvo, providing a comprehensive basis to assist with decision-making and assess the risks, constraints and preliminary costs of eradication in the whole island.

Here we present the results of monitoring exotic mammals from Corvo Island between 2009 and 2011 and investigate the feasibility of performing an eradication programme of mammalian species. To assess whether eradication is technically feasible, we proceeded in two steps. Firstly, the relative abundance and distribution of rodents, goats and sheep was determined every month and a sterilisation campaign was developed for cats. Specifically, we examined the hypothesis that exotic mammals vary in abundance responding to seasons and habitats, and that times with lowest abundance indices would be most effective for eradication. Secondly, the detected abundance values were compared with those from other islands where eradication was successfully executed to assess whether mammals with abundance ranges similar to those on Corvo have been successfully eradicated from other islands in the past. Finally, a revision of the actions needed to overcome any risks and constraints identified to attempt the eradication of introduced mammals on Corvo, was conducted.

METHODS

» Study area

Corvo (39º 40' N, 31º 7' W), an island of volcanic origin, has an area of 1700 ha - most of which is surrounded by steep cliffs >200 m in height (maximum elevation is 718 m). The island was inhabited in approximately 1558 (Chagas 1645-1650) (Fig. 1). The invasion of exotic mammal species occurred in the 15th century, facilitated by Portuguese settlers (Le Grand 1983; Monteiro et al. 1996). Of a total of ten terrestrial mammals introduced in the Azorean Archipelago, Corvo has five species that exist with feral populations (Table 2). They belong to three different taxonomic orders: Rodentia (black rat and house mouse *Mus domesticus*), Carnivora (feral cat) and Artiodactyla (goat *Capra aegagrus hircus* and sheep *Ovis aries*). On Corvo, one amphibian species, the Iberian water frog *Rana perezi* and one reptile species, the Madeiran lizard *Lacerta dugesii*, were also introduced. There are no native species of reptile, and only two native mammalian species are present on Corvo, the endemic Azorean bat *Nyctalus azoreum* and one *Pipistrellus* sp. bat. The native vegetation cover was almost exterminated (>90%) during the colonization process and, today, it is restricted to a few areas on the cliffs. Most areas were converted to agricultural fields and some others were forested with introduced plants (60%), mainly the African tamarisk *Tamarix africana,* Sweet pittosporum *Pittosporum undulatum* and Hydrangea *Hydrangea microphylla*. The remaining species (40%) are native to the Azores, *Erica azorica*, *Juniperus brevifolia*, *Picconia azorica*, *Morella faya*, *Laurus azorica*, *Vaccinium cylindraceum* and *Viburnum tinus*, and they are threatened through invasion of their habitats by non-native species and grazing by goats, sheep and cows.

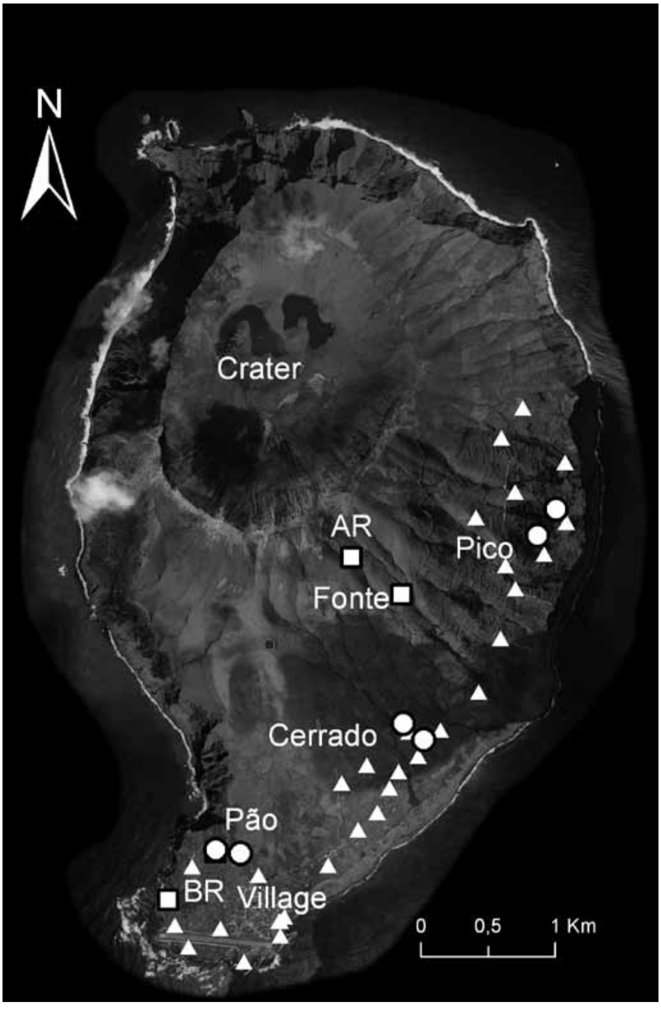

Figure 1. Spatial distribution of sampling areas for rodents (in white) at two different altitudes (squares) and in three different habitats (circles) and trapping areas for feral cats (triangles).

Figura 1. *Distribuição espacial das áreas amostradas para roedores (a branco) a duas altitudes diferentes (quadrados) e três habitats (círculos), e das áreas de captura de gatos assilvestrados (triângulos).*

Table 2. Occurrence of exotic mammalian species in the nine islands of the Archipelago of Azores.
Tabela 2. *Espécies exóticas de mamíferos presentes nas nove ilhas do arquipélago dos Açores.*

Exotic species	Corvo	Flores	Faial	Pico	São Jorge	Graciosa	Terceira	São Miguel	Santa Maria
Capra aegagrus hircus	X	X	X	X	X	X	X	X	X
Erinaceus europaeus			X				X	X	X
Felis catus	X	X	X	X	X	X	X	X	X
Mus domesticus	X	X	X	X	X	X	X	X	X
Mustela furo		X	X	X	X		X	X	X
Mustela nivalis			X				X	X	
Oryctolagus cuniculus		X	X	X	X	X	X	X	X
Ovis aries	X	X	X	X	X	X	X	X	X
Rattus norvegicus		X	X	X	X	X	X	X	X
Rattus rattus	X	X	X	X	X	X	X	X	X

» Monitoring the exotic mammalian species

Rodents

The accidental introduction of rodents was referred since the human colonisation in 1558 (Fructuoso 1591; Chagas 1645-1650; Cordeiro 1717). During the present study we identified the presence of two rodent species on Corvo, the black rat and house mouse.

The abundance index of rodents was assessed by live-traps on trapping grids for two years (2010 – 2012). During the first year, we aimed to assess differences in abundance between altitudes and identify temporal peaks in abundance. From March 2010 to February 2011, four areas were surveyed up to 10 times each (except in July and December because we were not on the island) once per month to identify abundance peaks throughout the year at two different altitudes: two grids < 250 m above sea level (asl), the Biological Reserve (BR), which is a predator proof enclosure area created by the LIFE project, and *Pão de Açúcar* (*Pão*), and two grids > 400 m asl, *Fonte da Lomba da Rosada* (*Fonte*) and the Altitude Reserve (AR), an area used by the LIFE project to restore native vegetation (Fig. 1). The trapping grids were 70 x 70 m and contained a total of 49 traps at 10 m spacing. The 49 traps were divided into 25 Sherman traps (for black rat and house mouse) and 24 Pest-stop traps (for house mouse only), which were alternated with each other throughout the grid.

During the second year, we established different study plots at similar elevation to examine differences among habitats. From September 2011 - March 2012 (5 monthly surveys, we were not able to survey the grids in October and February) we surveyed the three different habitats of Corvo: wooded riverbank in *Pico João de Moura* (*Pico*), pasture grassland in the *Cerrado das Vacas* (*Cerrado*) and urbane land in *Pão* (Fig. 1). Two 30 x 30 m grids each with 16 Sherman traps and 16 Pest-stop spaced 10 m apart were established in each habitat. Grids were 200 m apart within each habitat.

Traps were baited with peanut butter and operated for four nights each month. All captured individuals were marked. Because marks made by cutting the ear or hair were difficult to find in recaptures (most individuals had bitten ears and wet hair), animals were marked cutting a combination of a small part of their fingers. After handling, all individuals were released in their capture locations.

To compare rodent activity over time and between habitats we used an abundance index for house mice and black rats that was calculated using the number of individuals caught during each trapping session (4 nights) (Ruscoe et al. 2001). The number of individuals for each species was divided by the total number of trap-nights, corrected for traps that were sprung and thus unavailable, and expressed as the number of individuals captured per 100 trap nights (ind/100TN) (Cunningham & Moors 1993). For each month, we report the average trapping index and standard deviation for all grids that were in the same habitat type or at the same elevation.

Because our goal was to assess whether eradication was feasible, we also estimated rodent abundance using mark-recapture techniques (Otis et al. 1978; Ruscoe et al. 2001). We used our monthly trapping sessions as primary occasions, and each trapping night as secondary occasion in a robust design approach to estimate rodent population size for each month (Pollock 1982; Kendall et al. 1995). We accounted for individual heterogeneity in capture probability (Krebs et al. 1994; Conn et al. 2006), and evaluated models that varied capture probabilities over time and between habitats. We fitted models using Program MARK (White & Burnham 1999) via the RMark library (Laake & Rexstad 2008) in R 2.13.1 (R Development Core Team 2010) using the 'RDHet' model type. We selected the most parsimonious re-capture model structure using AIC, and present abundance estimates and 95% confidence intervals for each month from the most parsimonious model.

<u>Cats</u>

Cats were introduced on Corvo after 1717 (Cordeiro 1717), and once they were intentionally released to control rodent populations in pasture lands they became feral. On Corvo there are domestic, stray (cats fed by humans) and feral cats, but here we use the term 'feral' to refer to both populations, stray and feral ones.

The number of feral cats on Corvo was estimated at 163 individuals in 2011 using camera traps and spatially explicit capture-recapture models (Oppel et al. 2012). Here, we determined the trapping rate of feral cats to assess the amount of effort that would be required in a future eradication of cats on Corvo. Moreover, we developed a sterilisation programme of domestic cats to assess their number, and thus to evaluate the participation of cat owners to prevent the re-establishment of a feral cat population after their potential eradication.

In 2010, cat owners were individually contacted by house visits in a public awareness campaign. Domestic cats were micro-chipped and neutered with their owners' permission. This sterilisation programme was supported by the LIFE project and the local veterinarian, and was free of charge for owners. The acceptance rate of this sterilisation scheme was 90%, leading to the sterilisation of many domestic cats in 2010.

To assess the effort required for feral cat trapping, traps were deployed in areas known to be frequented by cats. From a previous study to assess their diet on Corvo, we knew that cats explored all habitats (*unpublished data*), and that higher densities were observed in the lower areas of the island (Oppel et al. 2012). Feral cats were captured using two Eezicatch and four Eeziset traps each night, in 56 trapping nights from February to August 2010. Traps were baited with fish after sunrise and then checked early in the morning. At the veterinary clinic, captured individuals were anesthetized, neutered and marked by cutting ¼ inch of their left ear. Once the cats had regained consciousness they were released in their captured locations.

<u>Goats and sheep</u>

Goats and sheep were introduced on Corvo along with the first inhabitants, to serve as a food resource (Fructuoso 1591). Wool production was an important activity until the 1960's, when meat production using bovines increased in popularity, and goats and sheep were restricted to inaccessible areas (coastline cliffs) of the island where they have established feral populations. Because their restricted distribution, transects by boat were conducted monthly along the coast of the island from September 2009 to August 2010, to estimate their number. During each survey, the entire island was circumnavigated and the cliffs were scanned with 10 x 40 binoculars to detect any goats and sheep. The number of all sheep and goats observed during a survey were summed to estimate an index of abundance for these species on the island of Corvo.

In addition, in November 2010 we conducted interviews with local farmers (n = 40) in order to ascertain the distribution and number of goats and sheep in the interior area of the island.

RESULTS

» Rodents

A total of 691 house mice and 44 ship rats were captured. Rodent abundance varied substantially over time, and was generally lower at higher elevation (Fig. 2). In the first year of monitoring, the highest abundance indices of house mice (13.5 ± 9.7 ind/100TN) and black rats (4.1 ± 5.8) were reached in November 2010 (Fig. 2). The lowest index of house mouse occurred from April to June 2010 (3.1 ± 0.7) and we did not capture any black rat in April and October 2010 and in January and February 2011.

Due to the low number of captures and recaptures of rats we were not able to estimate their abundance. The abundance estimation of house mice was limited to a grid at lower elevation, because few mice were captured and recaptured at the higher elevation site (Fig. 2). The temporal fluctuation in estimated abundance resembled the temporal fluctuation in the trapping index. The most parsimonious abundance model included a mixture component for capture probability that indicated heterogeneity in capture probability between 0.85 (95% CI: 0.21 - 0.99) for 5% of individuals and 0.28 (0.21 - 0.37) for the remaining individuals. Mouse abundance estimation for the grid covering approximately 0,5 ha ranged from 9 (7 - 17) individuals in May and June 2010 to 52 (45 - 66) individuals in November 2010 (Fig. 3).

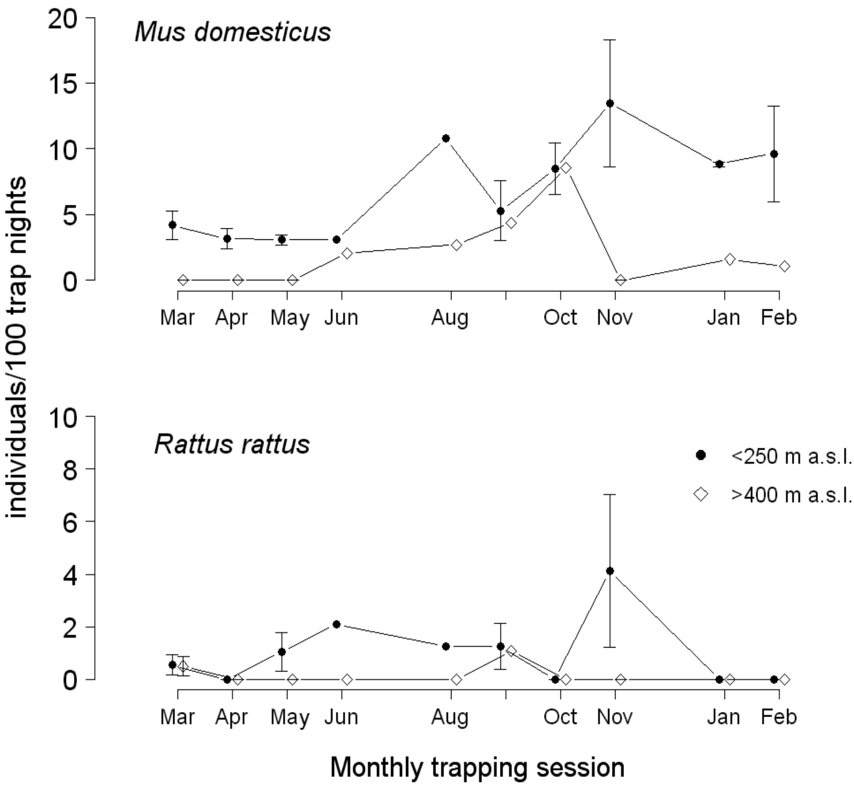

Figure 2. Mean trapping index (individuals captured per 100 trap nights) of house mouse *Mus domesticus* and black rat *Rattus rattus* captured on four grids in pasture grassland at two different altitudes from March 2010 to February 2011, on Corvo Island. Error bars represent standard deviation of mean trapping index across the two grids at each elevation and lack of error bars indicates that only one grid was operated in a given month.

Figura 2. *Índice médio de captura (indivíduos capturados por 100 noites de armadilhagem), de rato-caseiro* Mus domesticus *e rato-preto* Rattus rattus*, obtido em 4 grelhas com pastagem a duas altitudes diferentes de março a fevereiro de 2011, na Ilha do Corvo. As barras de erro representam o desvio padrão do índice médio de captura nas duas grelhas a diferentes altitudes e a ausência de barras de erro indica que foi amostrada uma única grelha nesse mês.*

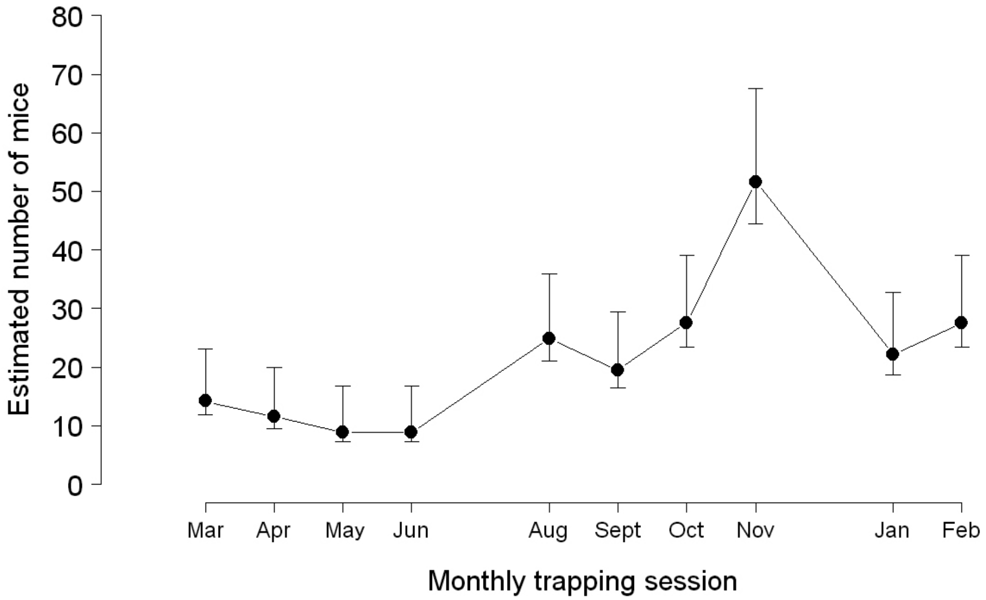

Figure 3. Estimated abundance (± 95% confidence intervals) of house mouse *Mus domesticus* in 0.5 ha of pasture grassland between March 2010 and February 2011 on Corvo Island.
Figura 3. *Abundância de rato-caseiro* Mus domesticus *(± 95% intervalo de confiança) estimada em 0,5 ha de pastagem, desde março de 2010 a fevereiro de 2011 na Ilha do Corvo.*

The comparison among habitats indicated similar temporal trajectories across the three studied habitats, but with lower amplitude in wooded riverbanks (range 12.3 - 16.8 ind./100TN) than in pasture grassland (range 6.3 - 22.8, Fig. 4). The highest abundance index of house mouse (22.8 ± 3.7) was obtained in pasture grassland in November 2011, and the lowest index occurred in urban land (5.2 ± 2.5) in March 2012. The most parsimonious abundance model included temporally varying capture probability that ranged from 0.14 (95% CI: 0.10 - 0.19) in September 2011 to 0.30 (0.23 - 0.38) in March 2012. Mouse abundance estimation for two grids covering approximately 0,4 ha ranged from 105 (86 - 137) individuals in pasture grassland in November 2011 to 17 (14 - 23) individuals in urban land in March 2012 (Fig. 5).

The highest abundance index of black rat was in wooded riverbank (11.4 ± 6.7 ind./100TN) in September 2011, but we did not capture any black rat in this habitat in December 2011 and January 2012, or in pasture grassland in November 2011, January, and March 2012 (Fig. 4).

» Cats

Feral cats were present in all studied habitats. During 336 trap nights, a total of 60 feral cats were captured, and only 4 of them were recaptured, leading to a total capture success rate of 0.08 cats per trap night. No domestic cats were captured in cat traps.

A total of 113 domestic cats were identified during the house visits. Of these domestic cats, the local veterinarian had already neutered 19% and most of them were micro-chipped. During our project an additional 51% were neutered and tagged between January and August 2010. Less than 10% of the owners (but owning 30% of total domestic cats) did not approve of sterilisation and a small minority did not approve that their cats be equipped with a microchip.

» Goats and sheep

The maximum number of 92 sheep was counted in January 2010, but other surveys had often substantially lower numbers (range 4 - 45), indicating that even the maximum count may have

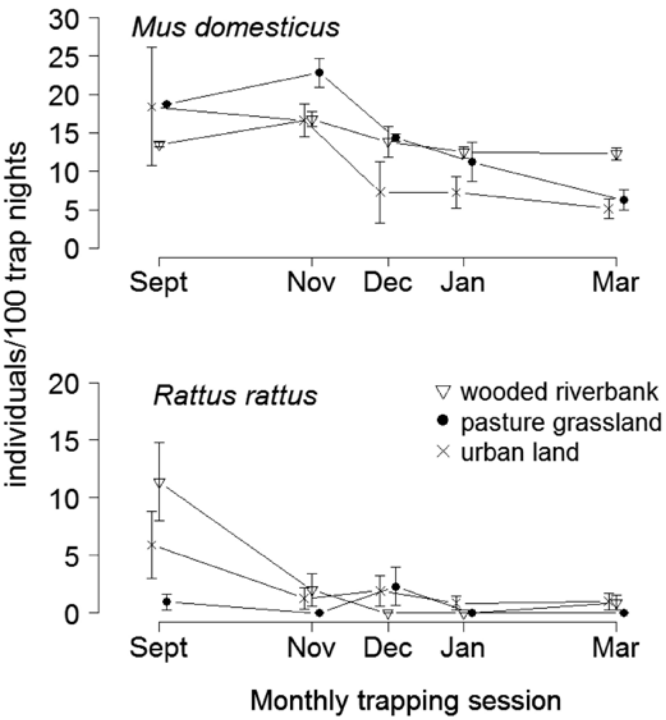

Figure 4. Mean trapping index (individuals captured per 100 trap nights) of House mouse *Mus domesticus* and Black rat *Rattus rattus* in three different habitats: wooded riverbank, pasture grassland and urban land between September 2011 and March 2012 on Corvo Island. Error bars represent standard deviation of mean trapping index across two grids in each habitat.

Figura 4. *Índice médio de captura (indivíduos capturados por 100 noites de armadilhagem) de rato-caseiro* Mus domesticus *e rato-preto* Rattus rattus *em três habitats diferentes: ribeira arborizada, pastagem e urbano, de setembro de 2011 a março de 2012 na Ilha do Corvo. As barras de erro representam o desvio padrão do índice médio de captura nas duas grelhas de cada habitat.*

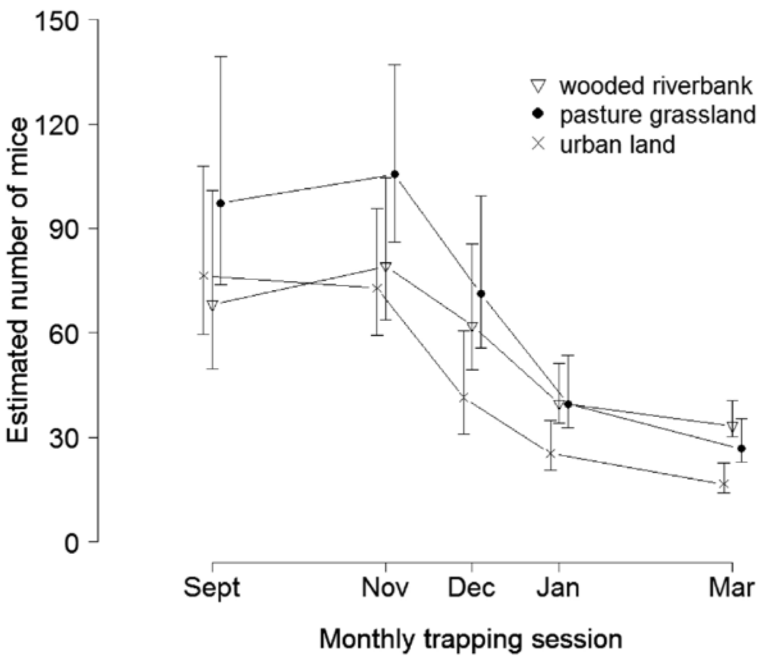

Figure 5. Estimated abundance (± 95% confidence intervals) of House mouse *Mus domesticus* in three different habitats: wooded riverbank, pasture grassland and urban land between September 2011 and March 2012 on Corvo Island. In each habitat, 2 grids were sampled that together encompassed an area of approximately 0.4 ha.

Figura 5. *Abundância de rato-caseiro* Mus domesticus *(± 95% intervalo de confiança) estimada em três habitats diferentes: ribeira arborizada, pastagem e urbano, de setembro de 2011 a março de 2012 na Ilha do Corvo. Em cada habitat foram amostradas duas grelhas, abrangendo no total uma área aproximada de 0,4 ha.*

missed some individuals. Likewise, the maximum count of goats was 93 in August 2010, and other surveys also did not detect substantial proportions of goats (range 25 – 90). Both sheep (65%) and goats (64%) were mainly distributed on the west side of the island.

Livestock data from farmers (50 – 100 sheep along the cliffs) matched the number obtained in our surveys. The number of goats and sheep observed in the interior part of the island varied between one to seven individuals, and they were mainly in the southwestern area.

DISCUSSION

» Is invasive mammal eradication justified and feasible?

<u>Rodents</u>

Based on our results, the eradication of rodents on Corvo should be attempted in April when the lowest peak in abundance occurred for both species. We found only minor differences in abundance indices or estimated abundance between different habitats, suggesting that equally high effort would be required in all habitats. However, the lower amplitude of temporal mouse abundance fluctuations in wooded riverbank habitat suggests that either food is more consistently available (e.g. trees bearing fruit throughout the year), or that sufficient shelter protects mouse populations from severe weather. Therefore wooded riverbanks on Corvo may require the most ground-based effort during a rodent eradication. We captured pregnant or lactating female mice in all sampled months, thus suggesting that mice breed all year around on Corvo, and that would also complicate a successful eradication.

Black rats with higher abundance index than on Corvo have been eradicated from a variation of island size, e.g. Buck Island with 59.3 ind./100TN (Witmer et al. 2007) and Bird Island with 141 ind./100TN (Merton et al. 2002). However, the design of our trapping grids may have been too small to be comparable to trapping indices from other studies. Our trapping grids were only 70 x 70 m in size, and may have therefore harboured only 1 – 3 rat home ranges, thus limiting the number of rats available for capture. The low abundance indices of black rats reported in this study must therefore be interpreted with caution; nonetheless, if the density of black rats on Corvo exceeded the density of rats on other inhabited islands from where these species were successfully eradicated (e.g. Great Barrier Island Ogden & Gilbert 2009), we would have expected much higher densities even in our very small trapping grid. Thus, we cautiously conclude that rat eradication is likely to be technically feasible.

Mice are harder than rats to eradicate from islands because of their movement behaviour and small home range sizes (MacKay et al. 2011), dietary neophobia, reduced access to bait where rats co-occur, and toxin resistance (Howald et al. 2007). House mice can reach higher densities (Howald et al. 2007) and successes and failures of mouse removal have occurred across the full range of island sizes and some eradications required more than one attempt (MacKay et al. 2007). In the Macaronesian islands, the only successful eradication of house mice was in the Selvagens Islands (200 ha) (Oliveira et al. 2008).Therefore, house mouse eradications require more meticulous planning than rat eradications (Martins et al. 2006). Factors such as bait application method, toxicant (first or second generation anticoagulant), and the presence of other introduced mammals should be taken into account for eradication attempts of both black rat and house mouse. Nevertheless, gaps in poison coverage (Micol & Jouventin 2002), low attractiveness of bait resulting in non-consumption despite encounter (Humphries et al. 2000), or resistance to the toxin used (Billing 2000) are more common reason for eradication failure of mice. MacKay et al. (2011) collected information about the population and individual behaviour of mice prior to their successful eradication from Saddle Island (New Zealand).

Our study indicated substantial fluctuations in house mouse abundance over the course of our study, and individual heterogeneity in the probability of capture. Such heterogeneity is well known in mice (Krebs et al. 1994), and would complicate any eradication, because for eradication to succeed every single individual must be removed regardless of its probability to enter a trap or consume bait. In order to increase the likelihood of success for house mouse and black rat eradication on Corvo, hand-spreading bait in conjunction with aerial drop and the use of at least two different toxins should be taken into consideration in the operational plan. Most people on Corvo are supportive of a rodent eradication campaign and recognise the potential advantages of a rodent-free status to the island's economy. Rodent populations on Corvo have been controlled for decades, using Difenacoum since 2006 (average of 500 kg per year). Because individuals could become resistant to certain toxins previously used on Corvo, the efficacy of different toxins should be tested before the bait for eradication of house mouse and black rat is chosen.

Black rats have higher impact on seabird population than brown rat *Rattus norvegicus* and Pacific rat *Rattus exulans* and prey mainly on burrow-nesting species (Jones et al. 2008). On three of the Macaronesian archipelagos (the Azores, Madeira and the Canaries), rodents mainly affect seabird species, although their effects have been observed as well on terrestrial bird and plant species in Madeira (Oliveira 2008) and the Canaries (Nogales et al. 2006; Traveset et al. 2009). On Corvo, eggs and chicks of Cory's shearwaters are relatively important in the diet of Black rat and House mouse (*unpublished data*). Rats have some impact on Cory's shearwater breeding success on Corvo (Hervías et al. 2012), on several islands in the Mediterranean (Lavezzi Island, Thibault 1995; Chafarinas Island, Igual et al. 2006) and in the Atlantic Ocean (Berlenga Island, Granadeiro 1991). All other seabird species breeding on Corvo are smaller than Cory's shearwater, breed in burrows and are likely vulnerable to both rats and mice; especially the two smallest seabirds, little shearwater and Madeiran storm petrel. When food resources are scarce for rodents, black rats can prey on these burrowing seabird species at all life stages (Jones et al. 2008). For these reasons, measures to control the population of rodents on this island is justified and could generate benefits for the entire island biodiversity including native plants (Meyer & Butaud 2009), mammals (Harris 2009), birds (Towns 2009) and invertebrates (Angel et al. 2009).

Cats

Cats were distributed among all habitats we studied on Corvo, which is in agreement with their generalist and opportunist trophic behaviour (Fitzgerald & Karl 1979; Liberg 1980; Natoli 1985; Barratt 1997; Alterio et al. 1998; Edwards et al. 2002), and has also been observed in the Canaries where a study on habitat use was conducted (Medina & Nogales 2007). On Corvo, as well as many other insular environments successfully colonized by cats (Van Aarde & Skinner 1981), they occupy the top of the terrestrial food chain. Feral cat eradications have succeeded on 83 islands in all oceans (Campbell et al. 2011). On the Canary Islands, two successful feral cat eradications have been carried out in two small non-permanently inhabited islets (Alegranza 10.2 km^2, and Lobos 4.4 km^2) where some important seabird colonies have been seriously affected by this predator (Ardura & Calabuig 1993; Rodríguez Luengo & Calabuig 1993; Martín et al. 2002a; Martín et al. 2002b). The biggest island where cat eradication has been successful is Marion Island, with a surface area of 290 km^2 (Campbell et al. 2011), and feral cats have also been eradicated from islands permanently inhabited by >300 people (Ratcliffe et al. 2010). Higher numbers of feral cats than those estimated in April 2011 for Corvo (range 123 - 228 individuals, Oppel et al. 2012) have been eradicated on other islands (e.g. Ascension, Macquarie, Marion, and Great Dog; Nogales et al. 2004; Ratcliffe et al. 2010). Judging from previous accomplishments, we suggest that a successful eradication of feral cats from Corvo Island (17 km^2 and 400 citizens) is technically feasible. However, domestic cats are currently not being sterilised and releases of unwanted kittens from domestic cats are not being prevented; therefore, the eradication of feral cats on Corvo is currently not feasible due to the almost certain re-establishment of a feral population from unwanted domestic progeny. Moreover, we found that some human cat lovers provide an excess of food around the village for feral cats. We did not find opposition to reduce the number of feral cats through sterilisation, but a few owners controlling 30% of domestic cats did not support the sterilisation of their cats.

Cats are generalist predators whose impacts have been reported in 120 different islands and on 175 vertebrates (Medina et al. 2011). In Macaronesia, particularly in the Canary Islands, cats prey on 68 species; of these, three giant lizards, three land-birds and two seabirds are threatened either globally or locally (Medina & Nogales 2009). On Corvo, cats prey upon several native Passeriformes (*unpublished data*) and accounted for an average of 80% of Cory's shearwater nest failures from 2009 to 2011 (Hervías et al. 2012). We were unable to monitor breeding success of other seabird species than Cory's shearwater because of their inaccessible nests; however during our project we found three recently fledged little shearwaters with incisor-marks of predation, which suggests that these species may currently be threatened by cats or restricted to breeding areas that are inaccessible to cats. For these reasons, the control of domestic cats and the eradication of feral populations are biologically justified.

We believe that eradication of feral cats would only be feasible after public education campaigns, as well as the establishment and enforcement of policies against feeding feral cats. Unfortunately, the main constraints we found are the lack of responsibility and political will by authorities to pass or enforce any legislation relating to cats, which renders a successful eradication highly unlikely at this time. As a first step, Corvo would need a veterinarian to record and sterilise all domestic cats and to keep an updated record of all domestic cats on the island.

Goats and sheep

Successful eradication programmes have been performed on different types of islands, ranging from 1 to > 100,000 ha (Campbell & Donlan 2005), with a high number of goats removed (> 40,000 on Pinta Island, Galápagos) (Campbell et al. 2004). So, in the case of goats and sheep on Corvo Island, where a total of 246 animals were counted, their eradication could be attempted, although due to the rough terrain conditions and the steep cliffs it can prove to be very expensive and may require aerial support to hunt and shoot remaining goats hiding in inaccessible places. Although goats and sheep are not of high economic importance for the island, some people enjoy shooting and eating these animals and may be opposed to eradication.

These herbivorous species affect not only endemic plants by predation and, consequently, their associated invertebrate fauna (Desender et al. 1999), they also destroy habitat for endemic and endangered seabird species such as Zino's petrel *Pterodroma madeira*, Fea's petrel *Pterodroma feae*, Cory's shearwater, Bulwer's petrels *Bulweria bulwerii* and Madeiran storm petrel (Oliveira 2008). For some of these reasons, goats were eradicated from Desertas Islands (Oliveira 2008). Therefore, although assessment of the impacts of goats and sheep on native species has not yet been attempted on Corvo, their pernicious effects on native and endangered plant species have been mentioned from other Macaronesian islands (Nogales et al. 2006).

» What actions are needed to overcome risks and constraints and to attempt mammal eradication on Corvo?

Our study of exotic mammals suggests that, according to the small size of the island and the densities of exotic species, their eradication should be technically feasible (Courchamp et al. 2003; Genovesi 2005). However, there are socio-political factors impeding the success of eradication attempts. There are at least three constraints to perform an effective eradication (Myers et al. 2000; Simberloff 2001): (1) Socio-political factors. Community opposition (Genovesi 2005) or releases of new exotic individuals from captive populations (Bomford & O'Brien 1995) are obstacles to eradication and will need to be solved before attempting an eradication. Any eradication on Corvo would be complicated by an inadequate legal basis, as regional laws do not regulate exotic species nor mitigate their impacts. The airport and harbor would need bio-security measures legally enforced to guarantee a successful implementation and therefore prevent new introductions (Oppel et al. 2011). Moreover, there are no measures to prevent immigrations and releases of new exotic individuals from existing captive populations. Education programmes must be carried out to inform the human inhabitants about the problems caused by exotic mammals for endangered species. On Corvo, good success has been achieved with the younger generation, but the older ones that makes up the majority of local government authorities, currently do not see the eradication of invasive species as a political priority. (2) Protection of non-target species will need to be implemented: Several native birds are likely to be at risk directly by eating poison bait if an open distribution was chosen for eradication. Azores wood pigeon, common chaffinch *Fringilla coelebs*, Atlantic canary *Serinus canaria*, blackbird *Turdus merula azorica*, common starling *Sturnus vulgaris granti* and yellow-legged gull. No other significant risks to non-target wild species have currently been identified. Among domestic animals, cows are the most significant non-target animals, because they occupy all parts of the island except cliffs, and are of economic importance to the inhabitants. Minimising accidental consumption of bait by cows is the greatest logistical challenge for a full rodent eradication on Corvo. The use of carefully selected methods for each exotic species (trapping, poisoning and shooting), along with some mitigations measures (antidotes, restricted areas for baiting and temporary captive populations), should be sufficient to avoid irreversible effects on non-target species. (3) Food availability: The attempt to eradicate black rat, house mouse and cats from Corvo Island could be a difficult task, unless food availability in the village, around livestock farms and in the uncontrolled dump place could be reduced to zero. The year-round availability of alternative food for exotic mammals could potentially lead to poor acceptance rates of poison bait among rodents and thus reduce the likelihood of success of a rodent eradication campaign.

Finally, the remoteness and usually bad weather condition is an unmanageable constraint on Corvo. An island which is generally windy with a high incidence of fog and rain, which may cause logistical complications in an eradication campaign that relies on an aerial spread of poison bait.

CONCLUSION

From the results obtained in this study we can conclude that the eradication of all exotic mammalian species from Corvo is technically feasible. However, especially for rodents, goats and sheep, sufficient information should be gathered prior to eradication to ensure an operational implementation that maximises the chances of success. There are also several constraints that will have to be overcome, mainly the limits of legal responsibility of authorities for management of exotic species, the risks of reinvasion and the accidental consumption of bait by cows which are very high and difficult to manage.

In the case of eradication, we strongly recommend community involvement in meeting and project-implementation activities and community consultation as being essential to generate support. Inhabitants should be actively involved in the implementation phase of the project and an expert should train some local community members in eradication methodologies.

Because unexpected and undesired secondary effects are, in general, more likely to occur when ecosystems contain more than one invasive species, the alternatives to eradication should be taken into account. For example, the removal of cats can increase the impact of rodents on seabirds. The eradication of rodents alone could lead to prey-switching in feral cats and thus seriously affect alternative cat prey such as seabirds (Dumont et al. 2010; Hervías et al. 2012). Furthermore the removal of herbivorous species such as goats and sheep can lead to the invasion of exotic plants that, in the absence of browsing, are more competitive than native plants, leading to an explosion of such weeds (Zavaleta et al. 2001). A large-scale and well-planned eradication campaign of all exotic mammals simultaneously would avoid such complications, but will require years of preparation to garner community support (Ogden & Gilbert 2009; Oppel et al. 2011).

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The first translocation of Cory's Shearwater *Calonectris diomedea borealis*, and review of SOS Cagarro campaigns, on Corvo Island, Azores

A primeira translocação de crias de Cagarro *Calonectris diomedea borealis*, e revisão de campanhas SOS Cagarro, na Ilha do Corvo, Açores

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ABSTRACT - The Azorean population of Cory's Shearwaters *Calonectris diomedea borealis* is the largest in the world, but is being negatively affected by invasive alien species and loss of breeding habitat. Predation by cats *Felis catus* and rats *Rattus* spp., and artificial light-induced mortality, are among the most significant threats for this species. Mitigation strategies such as *SOS Cagarro* rescue campaigns, and translocation of chicks to a safe breeding area, have been implemented on Corvo Island. Principal objectives were to facilitate population recovery, and create a new colony within a reserve denoted by a predator-proof fence. Translocated chicks were hand-reared in artificial burrows, and fed an artificial diet of 'whole Sardines' *Sardina pilchardus* and Chub Mackerels *Scomber japonicus*. Fledging success for the translocation was 90%. Measurements were taken of juveniles from 2009-2012, and from 10 translocated and eight control individuals in 2012; potential differences in fledging weight and wing length between years, and among transfers and controls, were investigated. Overall, 2012 saw the lowest mean fledging weight, but wing length values were higher than in 2011. Preferential investment into wing development may occur as a developmental adaptation when food availability is limiting, to minimise time in the nest. Although fledging weights of translocated and control chicks were similar, transfers had greater wing length. This is possibly due to increased feeding frequency of hand-reared individuals (to mitigate the impacts of stress due to handling), allowing greater investment into wing development; conversely, for naturally-reared chicks, provisioning occurs more infrequently due to heterogeneous distribution of resources, with prolongued periods of fasting.

RESUMO - A população de Cagarro *Calonectris diomedea borealis* dos Açores é a maior do mundo, mas é afectada por espécies invasoras introduzidas e por perda de habitat de nidificação. A predação por gatos *Felis catus*, Ratos *Rattus sps*. ou a mortalidade induzida por luzes artificiais são as ameaças mais significativas para esta espécie. No Corvo foram tomadas algumas medidas de mitigação, como as campanhas de salvamento SOS Cagarro ou a translocação de juvenis para uma área segura. O principal objectivo destas medidas é aumentar a recuperação da população reprodutora e criar uma nova colónia reprodutora dentro de uma área protegida rodeada por uma vedação à prova de predadores. As aves translocadas foram alimentadas à mão em ninhos artificiais com uma dieta de Sardinhas inteiras *Sardina pilchardus* e Cavalas *Scomber japonicus*. As aves translocadas saíram dos ninhos com uma taxa de sucesso de 90%. Foram medidas

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crias entre 2009 e 2012, e 10 aves juvenis translocadas e 8 de controlo em 2012; foram analisadas diferenças potenciais de peso e comprimento de asa na saída do ninho entre os vários anos e antre aves translocadas e controlos. Em 2012 as aves registaram valores de peso mais baixos, mas um maior comprimento de asa que em 2011. O investimento no comprimento de asa pode ser uma adaptação evolutiva quando a disponibilidade alimentar é baixa para diminuir o tempo de permanência no ninho. Embora o peso tenha sido semelhante entre juvenis translocados e crias de controlo na saída do ninho, as crias translocadas tinham maior comprimento de asa. Isto pode dever-se a uma maior frequência de alimentação das aves alimentadas artificialmente (para mitigação dos impactos do *stress* de manuseamento), que permita um maior investimento no comprimento de asa; em oposição, as aves controlo foram naturalmente alimentadas de forma menos frequente, devido a uma distribuição heterogénea de recursos com períodos de jejum prolongados.

The Azores Archipelago is an important breeding ground for the Cory's Shearwater *Calonectris diomedea borealis*, with 75% of the world's population breeding here (Monteiro et al. 1996, Fontaine et al. 2011). However, the species is becoming increasingly threatened by several anthropogenic factors, and the population size of Cory's Shearwaters breeding in the Azores is estimated to have fallen by 43% between 1996 and 2001 (Fontaine et al. 2011). An important threat is loss of native and/or endemic vegetation, driven by land conversion for human development and agriculture, which resulted in the loss of habitat in breeding colonies (Monteiro et al. 1996). Introduced invasive species are the principal drivers of species extinctions on islands (Igual et al. 2005) - particularly rats *Rattus* spp. (which can prey on all age classes of seabird depending on their size; Genovesi 2005) and also cats *Felis catus*. Their impacts are most significant on islands, due to lower levels of behavioural and ecological adaptation of indigenous species to introduced predators (Keitt & Tershy 2003, Nogales et al. 2004, Bried et al. 2008). Burrow-nesting species such as shearwaters are among the most vulnerable to predation by invasive mammals; these are, in part, attributable to current estimates of breeding success of Cory's Shearwaters on Corvo being low (39%, n = 461; Hervías et al. 2013). Eradication is internationally-recognised as a key management tool in the control of introduced species, and, under the Convention on Biological Diversity, is considered to be the preferred option in cases where their prevention has failed (Genovesi 2005). Due to their isolation, islands can be ideal locations to attempt such strategies as eradication, and considerable effort has been focused here in the last two decades (Le Corre 2008, Capizzi et al. 2010).

A further threat includes urban light-induced mortality at fledging (Fontaine et al. 2011). Due to their nocturnal behaviour at nesting sites, Cory's Shearwaters are particularly sensitive to artificial lights (Le Corre et al. 2002, Rodríguez & Rodríguez 2009). Fledglings are attracted to and become disorientated by artificial lighting in residential areas, a phenomenon termed "fallout" (Rodríguez & Rodríguez 2009, Fontaine et al. 2011, Rodríguez et al. 2012).

Cory's Shearwater translocation

Chick translocation is a widely-implemented method of seabird restoration, used to complement traditional techniques such as invasive species control and habitat management (Gummer 2003, Jones & Kress 2012). Chicks are transferred from natal colonies to artificial nests, and hand-reared to fledging; in order to accurately monitor chick growth and development, it is important to compare transferred individuals with control chicks naturally reared in natal colonies. Translocation is a useful technique to improve population viability, since it allows restoration of extirpated (locally extinct) colonies, establishment of new colonies, and can facilitate population connectivity between sub-colonies (Lewison et al. 2012). It is a particularly effective strategy for species that exhibit natal site philopatry such as shearwaters; although this trait means also that parents cannot be translocated with their chicks, since they would simply abandon the artificial burrow and return to their natal colony (Bell et al. 2005; Jones & Kress 2012).

Little is known about the population and breeding dynamics of Cory's Shearwaters on Corvo Island, the smallest island in the Azores Archipelago and the western-most point of Europe. Since 2009, considerable research has been carried out to address this, under the LIFE07 NAT/P/000649 project entitled 'Safe Islands for Seabirds'. We have been studying the island's breeding colonies, and implementing mitigation strategies to enable populations to recover from introduced predators, habitat loss and light-induced mortality (through the fledgling rescue project *SOS Cagarro*). Furthermore, as part of the work carried out for the LIFE project, Corvo Island is the location of the first-attempted translocation programme for Cory's Shearwaters. Begun was October 2012, the principal aim of the programme is to successfully hand-rear translocated chicks to fledging, and, in the long term, for them to return to breed in managed habitat within the island's Biological Reserve. It is hoped that this will not only establish new breeding populations on the island, but also contribute to the development of successful translocation techniques required to meet conservation objectives for more vulnerable species (Miskelly et al. 2009). Weight and wing length values at fledging were used as measurements of fledgling quality (Ramos et al. 2003). Our hypotheses were that: (1) weight and wing length would be reduced in translocated relative to control chicks, due to stress associated with frequent handling, and hand-rearing using an artificial diet. (2) weight and wing length of fledglings rescued in *SOS Cagarro* would be significantly different between 2009-2012, due to changing oceanic conditions caused by global climate change (Lewison et al. 2012). For example a study by Xavier et al. (2011) found that the proportion of Blue Jack Mackerel *Trachurus picturatus* consumed by Cory's Shearwaters, varied according to sea surface temperatures. Blue Jack Mackerel constitute the majority of the diet of Cory's Shearwaters during the breeding season (Xavier et al. 2011); in addition to the Sardine *Sardina pilchardus* and Chub Mackerel *Scomber japonicus* used in the present translocation study, this species is fished commercially in the Azores, used as live bait for tuna fishing (Martin, 1986, cited in Xavier et al. 2011). It is therefore expected that varying annual intensities of prey exploitation by fisheries, could also potentially impact on resource availability and, consequently, fledgling weight and wing length (Xavier et al. 2011).

METHODS

» Study Site

Fieldwork and *SOS Cagarro* campaigns were carried out on Corvo Island, situated in the sub-tropical North Atlantic Ocean (39º 40' N, 31º 7' W) (Xavier et al. 2011). Corvo has a unique village (Vila do Corvo), and during the Cory's Shearwater fledging season, artificial lights are switched off around the island, except for in this residential area. For the translocation, artificial nests were built within the island's Biological Reserve, a ~3 ha enclosure site denoted by a predator-proof fence, which has undergone considerable management under the LIFE project. Burrows were fitted with drainage pipes to prevent flooding, and a gap to connect opposing, but not adjacent, burrows. The doorway was designed to be wide enough for chicks to enter and leave their nests, and a stick was placed at the entrance of each burrow to monitor pre-fledging emergence behaviour. Other habitat management strategies that have been implemented inside the Reserve include restoration of endemic plant species, and extensive control of invasive predators (see Action C3 in final report of LIFE project).

» SOS Cagarro

SOS Cagarro rescue campaigns have been implemented in the Azores since 1995 to reduce cases of light-induced mortality. On Corvo Island, the Portuguese Society for the Study of Birds (SPEA) have recorded fledgling weight measurements between 2009 and 2012, and wing length from 2011-2012. Once chicks began to leave their nests for the ocean at approximately three months of age (October, Ramos et al. 2003), nightly patrols of the village were carried out with local school children between 20:30 and midnight, to search for fledglings. Rescued individuals were fitted with identification rings, and weight and wing length measurements taken, to monitor population trends and, in future years, ascertain the proportion of individuals returning to breed.

The *SOS* rescue campaign is also proving to be a useful tool in environmental education and public engagement. It has been met with great enthusiasm from local school children, who enjoy being involved in rescuing fledglings and returning them to their natural habitat; furthermore, many local people and businesses on the island show continued support by reporting or rescuing fledglings found outside of nightly patrols.

» Translocation protocol

Fieldwork was carried out between 2nd October and 1st November 2012. Ten chicks were removed from natural burrows in three locations around the island. Nest selection considered ease-of-access, and whether the burrow entrance was concealed sufficiently to prevent early imprinting; also, chicks were selected with primary feathers half grown. Nest litter and shed chick down were collected from each burrow, to reduce stress associated with re-housing, and to facilitate olfactory recognition of the artificial nest after emergence and fledging ('locality imprinting', Serventy et al. 1989; cited in Gummer 2003). Each individual was ringed, and weight and wing length measurements were taken, before being transferred to a cardboard box and transported 30 minutes away, to artificial nests in the Biological Reserve. Chicks were not fed on the day of translocation, to prevent stress-induced vomiting which can cause suffocation (Hozumi et al. 2011). Three cameras were positioned approximately two metres away from the burrow entrances, to monitor chick emergence behaviour.

A further eight chicks were selected from four locations, to be used as controls with which growth measurements of translocated individuals could be compared. As before, each chick was ringed, and weight and wing length measurements were taken, before returning them to their nests. Weight and wing length were measured every four days to prevent undue stress which may inhibit development.

Although precise age of chicks was unknown, average hatching date on Corvo between 2009 and 2011 was 25th July 2012 (unpublished data), therefore chicks were estimated to be 10 weeks old. Date of fledging was assumed to be the last day on which the chick was found in its nest; fledging weight and wing length were assumed to be those values obtained on that day.

» Development of artificial diet

We attempted to replicate the natural diet of Cory's Shearwaters as closely as possible, alternating between Sardines (Granadeiro et al. 2000), and a combination of Chub Mackerel (Granadeiro et al. 1998) and Sardines. Feeding protocol differed slightly from the standard for translocations, using whole instead of tinned Sardines (Hozumi et al. 2011). Although the latter is known to yield 86-100% fledging success, there is now speculation that this diet does not provide the same nutritional balance as natural regurgitation from parents in the wild (Hozumi et al. 2011). Furthermore, the present study used solid, rather than puréed, sardines, because it is a more realistic interpretation of the natural feeding strategy; a second motivation behind this decision was based on previous experiences of individual authors on Corvo Island, where it was found to be easier to feed adult Cory's Shearwaters with solid sardines rather than the puréed form.

Since this was the first attempted translocation of Cory's Shearwaters, estimation of appropriate daily weight changes was based on previous translocations of medium-sized Fluttering Shearwaters *Puffinus gavia*; a minimum weight gain of 5-10g per day was decided upon (Hozumi et al. 2011), with each chick being fed (provisionally) between 60g and 70g per day. Fish were soaked in water prior to feeding, to replace water lost through freezing and defrosting processes, and to prevent dehydration of chicks. In order for chicks to become accustomed to the artificial diet, the first meal was diluted with 50% water, and subsequently with 25% water, in-keeping with feeding protocol used in a previous translocation of Fluttering Shearwaters by Gummer & Adams (2010, cited in Hozumi et al. 2011).

Prior to feeding, each chick was weighed, and wing length was measured, to monitor overall growth and development. Initially, chicks were fed daily between 9am and 10am, beginning the day after translocation to artificial nests. Once feeding began, the number of participants was kept to a minimum: one person to securely hold the chick whilst carefully extending the neck to open the oesophagus, and a second person to deliver the food, massage the throat to encourage swallowing, and clean the chick with a towel to prevent soiling of the feathers (Miskelly et al. 2009).

Since wing length is considered to be an effective predictor of time until fledging (Hozumi et al. 2011), provisioning of chicks ended once they reached a wing length of 35.0cm or greater, to encourage them to abandon the nest. This reflects a similar provisioning strategy to that adopted in natural burrows, whereby parental investment gradually decreases in the days preceding fledging, to encourage chicks to leave in search of food (for example see Gangloff & Wilson 2004). As the first attempted translocation of Cory's Shearwaters, it was difficult to estimate ideal wing length at which provisioning should cease. The value of 35.0cm was selected based on mean natural fledgling wing length data recorded in *SOS Cagarro* rescue campaigns from 2011. Once feeding stopped, weight and wing length measurements were still taken on alternate days, to monitor development.

» Modifications to feeding protocol

For the first three days' feeding, fish were left to soak in cooled boiled water overnight prior to feeding. Although fish did absorb the water (percentage weight gain 0.52% – 0.92% for sardine diet; 2.41% for mixed diet), it caused them to break up, making feeding more difficult. Subsequently, fish were not soaked overnight, but instead injected with purified water (25% of dry weight of each fish) on the day of feeding.

Gradual but continued weight loss during the first six days of feeding resulted in modification of the diet. Each chick was fed between 90g and 100g daily (approximately 1.5 fish), and injected with 25ml of purified water (25% of the upper weight range, to account for water loss during defrosting). After three days, all 10 chicks increased in weight (between 10g and 50g).

Despite increased weight and wing length for each chick, it was decided that feeding should take place on alternate days, to reduce the likelihood of stress inhibiting weight gain. Initial measurements taken following one missed days' feeding, showed that all chicks lost weight (between 20g and 80g). However, wing length continued to increase for each individual, and all weight values recorded were between 890g and 1120g - within appropriate limits for fledging (approximately 900g, Mougin et al. 2000). Furthermore, this range is also higher than the average fledgling weight recorded during *SOS Cagarro* campaigns between 2009 and 2011 (785g). Since only one regurgitation occurred throughout the trial, it was decided that feeding solid fish should continue.

» Statistical analyses

Statistical analyses were conducted only for juveniles rescued in *SOS Cagarro* campaigns between 2009 and 2012. Weight and wing length data were analysed (individually) using one-way Analysis of Variance (ANOVA), to investigate whether juvenile condition at fledging was significantly different between 2009 and 2012. Residuals from the model output were then tested for normality using the Anderson-Darling test (library 'nortest').

Correlation was also used to test for a relationship between weight and wing length; this used data from 2011 and 2012 only, since wing length data were unavailable for 2009 and 2010. Firstly, a linear model was run in the form: wing length ~ weight. Residuals were then tested for normality using Anderson-Darling normality test, and a Pearson's product-moment correlation was carried out. All analyses were conducted using the statistical package 'R' version 2.9.2 (R Development Core Team 2009) with a significance level of $p < 0.05$. Results given are mean ± standard deviation (SD), unless otherwise stated. R-squared (R^2) values quoted refer to adjusted R^2 values in the model output.

RESULTS

» Translocation

Throughout the period of hand-feeding, chicks frequently changed artificial burrows, occasionally to burrows with another chick present in the adjoining nest. Wherever possible, chicks were returned to their original nests. Towards the end of the trial, as chicks approached fledging, emergence behaviour increased in translocated individuals. Camera traps revealed nocturnal movements outside of burrows such as flexing of the wings, calling, and occasional bill-to-bill contact with other emerged chicks. Excrement was also found immediately outside, and up to 1m away from, the burrows.

(i) Translocated Chicks

Overall fledging success for the translocation was 90%, with one fatality. This chick was possibly one of the youngest to be transferred, as it was one of the heavier individuals, and wing development was delayed in comparison to the other chicks. Despite this, it had shown no previous signs of ill-health; its death was thought to be due to a failed fledging attempt during a hurricane on 28th or 29th October, following which the chick was found almost dead outside the artificial nests. Although initially revived and replaced in the nest to dry its wings thoroughly, the chick later died, possibly from hypothermia or an infection.

Fledging occurred between 14th and 24th October, with the majority between 22nd and 24th October. Fledgling weight ranged between 730.0g and 1020.0g, with a mean value of 864.4g (SD = 87.9, median = 860.0g; Fig. 1, Table 1). All chicks except one had a wing length of 35.0cm or greater at fledging, ranging between 35.4cm and 37.8cm and with a mean value of 36.1cm (SD = 0.7, median = 36.1cm; Fig. 2, Table 1). The exception, LV04002, attempted to fledge at 34.5cm but failed due to bad weather conditions. This chick later died, and so final wing and weight measurements were omitted from subsequent data analyses.

(ii) Control Chicks

Overall fledging success for chicks reared in natal colonies was 100%. Fledging lasted from approximately 18th – 29th October, with the majority fledging between 18th and 25th October. Upon fledging, individual weights ranged between 690.0g

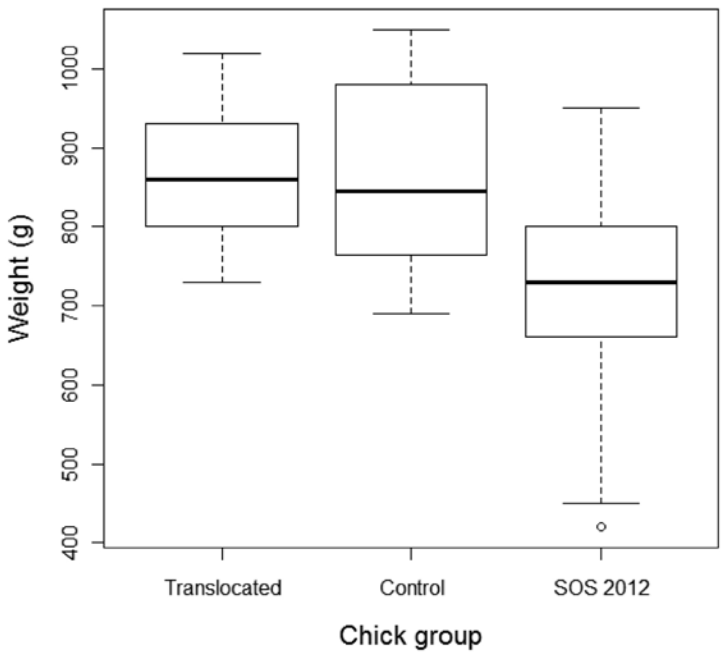

Figure 1. Weight recorded for translocated and control chicks, and juveniles rescued in the *SOS Cagarro* campaign 2012; median (central line), 25-75% inter-quartile range (box), non-outlier range (whiskers), and outlier values.

Figura 1. *Peso registado para aves juvenis translocadas, aves de controlo e aves juvenis recuperadas durante a campanha SOS Cagarro de 2012; mediana (linha central), variação inter-quartil 25-75% (caixa), variação sem valores extremos (bigodes) e valores extremos.*

and 1050.0g, with a mean value of 865.0g (SD = 136.6, median = 845g; Fig. 1, Table 1); wing length ranged between 33.7cm and 35.8cm, with a mean value of 34.9 (SD = 0.8, median = 35.0; Fig. 2, Table 1).

» SOS Cagarro Rescue Campaign

Overall, 2011 was a particularly heavy year for cases of 'fallout' (n = 227, Table 1); conversely, in 2012, fledging occurred over the shortest period of time, and fledgling frequency was the lowest recorded between 2009 and 2012, with just 98 chicks rescued (Table 1). Mean juvenile weight ranged from 723g to 800g (median = 730.0 - 800.0g; Fig. 3, Table 1). Although 2012 saw the lowest mean weight (723.0g, median = 730.0g; Fig. 3, Table 1), mean wing length was actually higher relative to 2011 (Table 1, Fig. 4; data unavailable for 2009-2010).

Mean juvenile weights were found to be significantly different between 2009 and 2012 (ANOVA, $F_{3,584} = 27.07$, $P < 0.001$, adjusted $R^2 = 0.12$), therefore *post hoc* pairwise comparisons were

Table 1. Weight and wing length measurements recorded for translocated and control chicks at fledging, and juveniles rescued in *SOS Cagarro* campaigns between 2009 and 2012.

Tabela 1. *Pesos e comprimentos de asa registados para crias translocadas e de controlo ao saírem do ninho e para crias capturadas nas campanhas SOS Cagarro entre 2009 e 2012.*

	Translocated	Control	2009	2010	2011	2012
Start date*	-	-	17.10	21.10	19.10	19.10
End date*	-	-	30.10	22.11	17.11	12.11
n	10**	8	160	193	227	98
n^1	-	-	147	190	169	82
n^2	-	-	-***	-***	46	65
Min weight (g)	730.0	690.0	500.0	440.0	500.0	420.0
Max weight (g)	1020.0	1050.0	1050.0	1050.0	920.0	950.0
Mean weight (g)	864.4	865.0	795.0	800.0	729.0	723.0
Median weight (g)	860.0	845.0	800.0	800.0	730.0	730.0
Standard deviation (weight)	87.9	136.6	90.7	102.4	84.4	103.6
Min wing length (cm)	35.4	33.7	-	-	34.0	31.2
Max wing length (cm)	37.8	35.8	-	-	37.4	38.5
Mean wing length (cm)	36.1	34.9	-	-	35.9	36.3
Median wing length (cm)	36.1	35.0	-	-	36.0	36.4
Standard deviation (wing length)	0.7	0.8	-	-	8.8	11.7

n = Total number of number of fledglings rescued during *SOS Cagarro*. (For translocated and control groups, 'n' refers to the total number of chicks used for the translocation study).
n^1 = Total number of chicks for which weight data were available.
n^2 = Total number of chicks for which both weight & wing length data were available.
* = Of annual *SOS Cagarro* campaigns.
** = Analyses of weight and wing length data are based on data obtained from nine translocated individuals; one fatality was omitted from analyses since it did not fledge.
*** = Wing length data unavailable for 2009 and 2010.

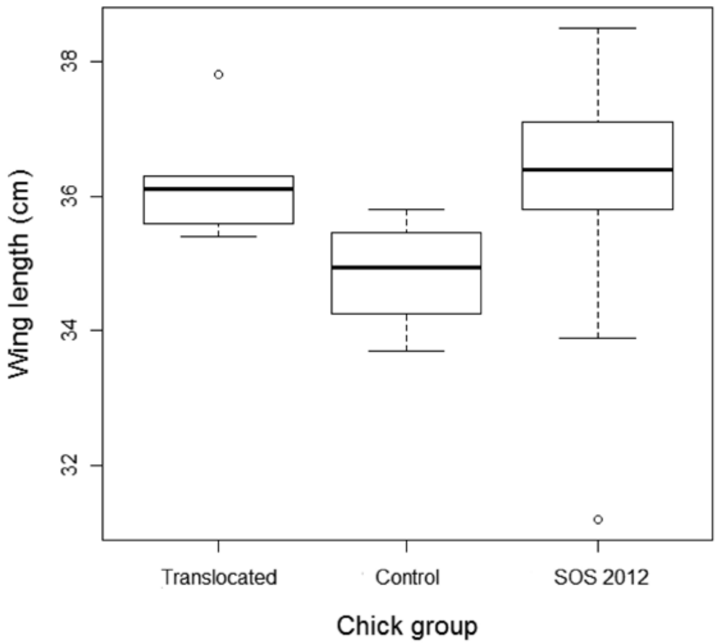

Figure 2. Wing length recorded for translocated and control chicks, and juveniles rescued in the *SOS Cagarro* campaign 2012; median (central line), 25-75% inter-quartile range (box), non-outlier range (whiskers), and outlier values.
Figura 2. *Comprimento de asa registado para aves juvenis translocadas, aves de controlo e aves juvenis recuperadas durante a campanha SOS Cagarro de 2012; mediana (linha central), variação inter-quartil 25-75% (caixa), variação sem valores extremos (bigodes) e valores extremos.*

carried out to highlight for which year groups this was most apparent (Dytham 2011).

Post hoc tests revealed a significant difference in mean fledgling weights between 2009/2011 (Tukey's HSD, $P < 0.001$), 2009/2012 (Tukey's HSD test, $P < 0.001$), 2010/2011 (Tukey's HSD test, $P < 0.001$), and 2010/2012 (Tukey's HSD test, $P < 0.001$). Interestingly, there was no significant difference in mean fledgling weight between 2011 and 2012, but a significant difference was found in mean wing length between these two years (ANOVA, $F_{1,109} = 4.80$, $P = 0.03$, adjusted $R^2 = 0.03$).

A highly significant, positive association was found between weight and wing length (Pearson's product-moment correlation, $r_{109} = 0.44$, $P < 0.001$, t = 5.10; Fig. 5). This result is based on data from 2011 and 2012, using individuals for whom both weight and wing length data were available (2011: n = 46; 2012: n = 65, Table 1).

DISCUSSION

» (i) SOS Cagarro 2009 - 2012

Based on data recorded for 2009-2012, the highest and lowest frequencies of rescued juveniles were observed in 2011 and 2012 respectively. This may be due to differences in the number of breeding adults, and also breeding success, which are heavily dependant on food availability and quality (Wanless et al. 2005).

In addition, mean juvenile weight was also lowest in 2012 relative to 2009-2011, suggesting a reduced rate of provisioning - possibly caused by decreased food availability. However, mean weight was not significantly different between 2011/2012. This suggests a secondary factor driving reduced fledgling frequency in 2012: possibly increased nest predation by introduced predators. Furthermore, the positive correlation found between (combined) weight and wing length data for 2011 and 2012 actually implies that increased provisioning allowed

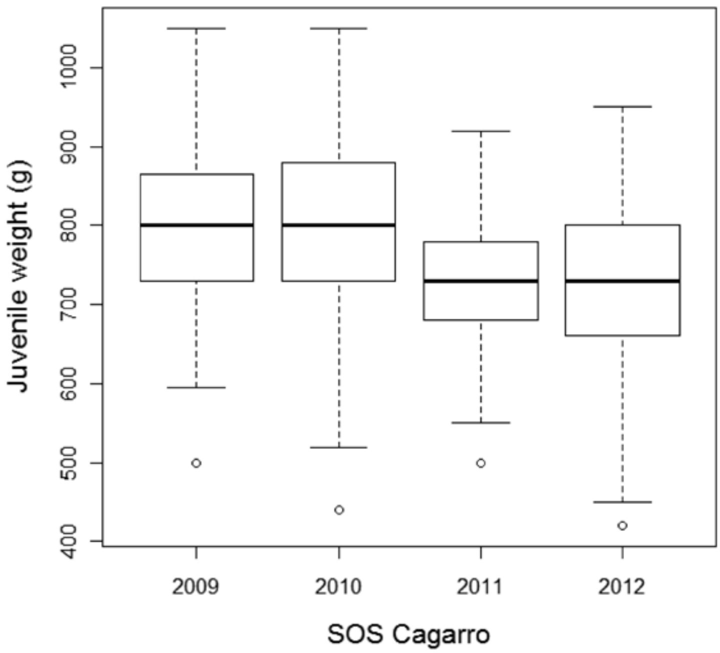

Figure 3. Juvenile weight recorded for individuals rescued in *SOS Cagarro* campaigns between 2009 and 2012; median (central line), 25-75% inter-quartile range (box), non-outlier range (whiskers), and outlier values.
Figura 3. *Peso registado para aves juvenis recuperadas durante as campanha SOS Cagarro entre 2009 e 2012; mediana (linha central), variação inter-quartil 25-75% (caixa), variação sem valores extremos (bigodes) e valores extremos.*

greater resource investment into wing development.

Despite 2012 showing decreased overall mean juvenile weight compared to 2009-2011, wing development did not appear to be compromised, and was actually significantly higher than the average recorded for 2011. A possible explanation for this may be preferential investment in wing development over body weight. According to O'yan & Anker-Nilssen (1996), in seabirds demonstrating low fecundity, chicks may allocate resources to certain body parts when food availability is limiting, to minimise time spent in the nest and dependency upon parents to feed. O'yan & Anker-Nilssen (1996) found that reducing food intake for the burrow-nesting Atlantic Puffin *Fratercula arctica* caused reduced growth rate (including body mass), however wing length grew preferentially relative to other characteristics measured. (Arguably, such developmental adaptations may compromise on the proportion of fat reserves allocated to oil production, required to increase permeability of feathers to water. Upon release, a few individuals rescued in *SOS Cagarro* 2012 appeared to experience difficulty swimming, which may be due to lack of oil secretion to, and subsequent water absorption by, the feathers.) Unfortunately, with no fish landing data available for 2009-2011, it is impossible to determine cause-and-effect relationships between prey availability, variation in weight and wing length, and subsequent fluctuations in fallout frequencies for the present study.

» (ii) Comparisons of weight and wing length from translocation and SOS Cagarro 2012

Fledging was slightly delayed for naturally-reared control chicks in comparison to transferred chicks, beginning and ending slightly later. A possible explanation for this occurrence could be that control chicks were slightly smaller. Natural chick provisioning occurs more infrequently compared to feeding of translocated chicks - which took place at regular intervals to offset the potential effects

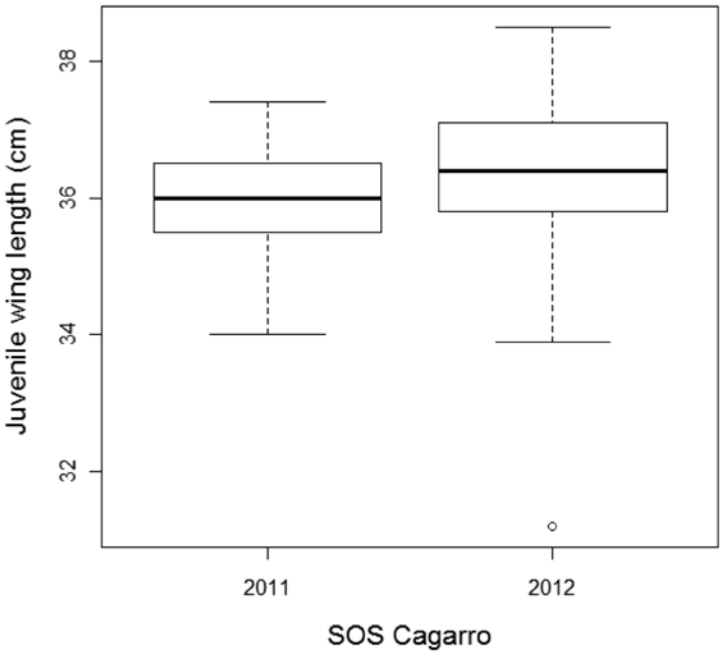

Figure 4. Juvenile wing length recorded for individuals rescued in *SOS Cagarro* campaigns between 2011 and 2012. (Note: data unavailable for 2009 and 2012); median (central line), 25-75% inter-quartile range (box), non-outlier range (whiskers), and outlier values.

Figura 4. *Comprimento de asa registado para aves juvenis recuperadas durante as campanhas SOS Cagarro de 2011 e 2012; mediana (linha central), variação inter-quartil 25-75% (caixa), variação sem valores extremos (bigodes) e valores extremos. (Nota: dados não disponíveis para 2009 e 2010).*

of stress, induced through frequent handling and hand-feeding. Procellariiformes (including shearwaters and storm petrels) are known to visit the colony infrequently, due to heterogeneity in the spatio-temporal distribution of resources; therefore it is not uncommon for Cory's Shearwater chicks to experience fasting periods of several days (Granadeiro et al. 2000; Gangloff & Wilson 2004). Despite this discrepancy in fledging period, the highest frequency of fledging events occurred within a similar time frame for both control and translocated groups; this, combined with similar mean fledging weights between these two groups, provides additional support for the integrity of the feeding protocol used in the translocation.

Mean weight recorded for *SOS* 2012 juveniles was considerably lower than that for hand-reared and control chicks. This is as expected, because weight values used for translocated and controls were the last recorded measurements of individuals in the nest, whereas weight values for *SOS* juveniles were recorded following considerable energy expenditure during the first flight.

Interestingly, there was considerable variation in mean fledgling wing length between translocated and control groups (although this could not be analysed statistically), and was found to be higher for hand-reared chicks. This may be linked to more frequent, regular feeding of translocated relative to (naturally-reared) control chicks, allowing more resources to be invested in wing growth. However, with this in mind, one might also have expected to observe (relatively) lower mean fledgling weight in control chicks, but was actually found to be similar. It is possible that an alternative form of energy expenditure caused (relatively) decreased wing length in control chicks – possibly investment into alternative morphological development, or

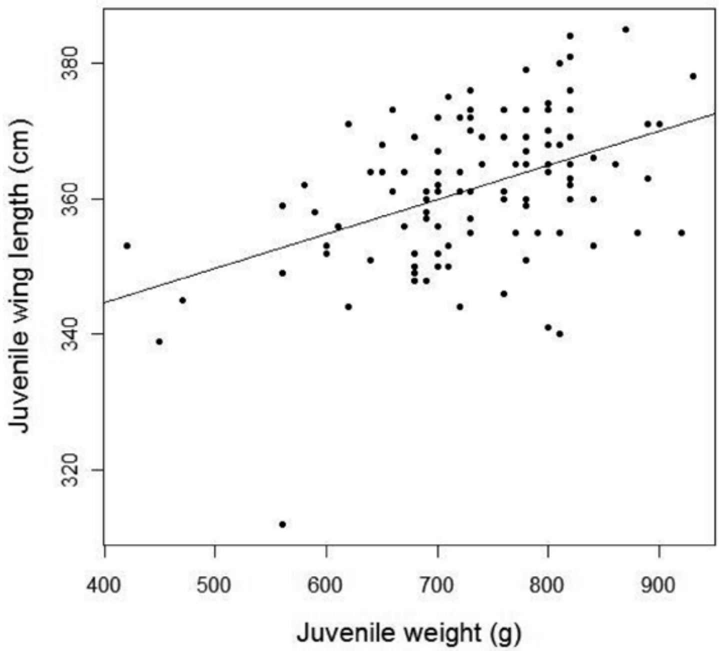

Figure 5. Correlation between weight and wing length, for juvenile Cory's Shearwaters rescued in *SOS Cagarro* between 2011 and 2012.
Figura 5. *Correlação entre peso e o comprimento de asa, para juvenis de Cagarro capturados nas campanhas SOS Cagarro entre 2011 e 2012.*

physiological or metabolic processes (for example maintenance of body temperature). However, if observed discrepancies in mean wing length were due to more frequent feeding of hand-reared relative to naturally-reared chicks, one might have also expected to find similar mean wing length between *SOS 2012* and control individuals. This was not the case; mean *SOS* 2012 wing length was actually almost equal to that for hand-reared chicks (although *SOS* 2012 values occupied a much larger range). Unfortunately, due to a small sample size for the translocation study (translocated n=9 and control n=8), it was not possible to investigate whether these findings were statistically significant.

» (iii) Translocation risk analysis

Impact of translocation on source populations

The present translocation study reflects a small-scale trial, using a species currently considered as Least Concern by the IUCN, due to its widespread geographic range (Birdlife International 2012). It is unlikely that the removal of 10 chicks, on an island whose total breeding population is estimated to be around 22,980 pairs (Fontaine et al. 2011), should impact negatively on the overall breeding dynamics of source populations. However, Gummer (2003) recommends a large source population to promote the success of a translocation; furthermore, should the trial on Corvo be extended in the future, a considerably larger sample size would be required to conduct statistical analyses. With these in mind, potential impacts on source populations should be assessed as a pre-requisite to future translocations, and ensure that chicks are selected from stable source populations – particularly on an island which is currently not considered to be 'predator free', and so population growth is vulnerable to other external pressures posed by the presence of cats and rodents.

Probability of return

As a highly philopatric species, Cory's Shearwaters lend themselves to be ideal study species for a translocation. However, this trait can also add complications to the methodology, and influence the long-term success of the programme. One of the most common reasons that translocations yield low adult return rates is that chicks are not transferred to artificial nests before imprinting of the natal burrow has taken place (Gummer & Adams 2010), which would increase the likelihood of transferred individuals returning to the site as breeding adults (for example see Serventy 1989, cited in Bell et al. 2005). To our knowledge, the age of imprinting in Cory's Shearwaters is unknown; however, according to discussion by Hozumi et al. (2011), Fluttering Shearwaters require a minimum of 10 days to imprint to a new nesting site. In the present study, chicks were transferred one month before anticipated fledging, to minimise the likelihood of imprinting being a factor influencing return rate.

» (iv) Translocation: lessons and further work

Being the first attempted translocation of Cory's Shearwater chicks, this presented some difficulties with regards to protocol design and implementation. For example ascertaining feeding frequency (which is irregular in the wild), appropriate values for daily weight gain, and the amount of fish required to achieve fledging weight. Despite these initial obstacles, comparisons between transferred and control chicks suggest that the artificial diet was an accurate replication of the natural diet.

Concerning improvements to the translocation procedure and post-transfer care, some studies have administered water to chicks after initial transfer to artificial nests, to reduce potential impacts of heat stress and dehydration (for example Miskelly et al. 2011). Once feeding begins, it is recommended that fish be cut diagonally rather than laterally, to keep the maximum width of the fish to a minimum, and facilitate swallowing.

Finally, to allow chicks to adjust to their new burrows, Hozumi et al. (2011; based on protocol used in Miskelly et al. 2009), recommends blocking the burrow entrance with plastic mesh for the first three days after translocation. This is potentially a useful technique to prevent premature chick disappearances, and ensure that appropriate fledging weights are reached (Gummer & Adams 2010). Although burrows were not completely blocked in the present study, it is uncertain whether it would have reduced the frequency of nest swapping incidences amongst chicks. In a translocation study of Fluttering Shearwaters, Gummer & Adams (2010) used burrow blockades, but also experienced high levels of burrow-swapping amongst chicks – including to burrows of the Fairy Prion *Pachyptila tutur*.

According to Bell et al. (2005), the most accurate measure of success for a translocation is the number of individuals who return to breed as adults. However, this requires post-translocation monitoring, which for species demonstrating delayed sexual maturity and low fecundity, is a long-term financial commitment. Concerning the immediate future, habitat management must be considered a priority; for species demonstrating natal site philopatry, habitat quality is a key factor in establishing new colonies (Jones & Kress 2012). Therefore, preventing reinvasions of invasive species is of particular importance, through maintenance of the predator-proof fence, and continued efforts in predator control and restoration of native vegetation (Hozumi et al. 2011).

Once returning adults are expected (for Cory's Shearwaters, this is in February, approximately 6-8 years after fledging), acoustic systems should be placed within the Reserve to attract returning birds. At this time, searches should also be implemented to identify recaptures, in order to ascertain recruitment rate (i.e. the number of transferred adults returning to breed). In a translocation study of Audouin's Gulls *Larus audouinii*, Oro et al. (2011) used capture-recapture modelling to analyse fledgling survival, and the probability of their returning to breed as adults. Information derived from such analyses can be used to inform translocation methodology for subsequent translocations, for example whether distance between natal and artificial burrows is influential in determining return rates. Whilst it would be interesting to carry out such analyses in future translocations, it is not possible for the present study since a considerably larger sample size is required.

The methodological and financial implications of the present study extend well beyond the time of expected first returning adults, highlighting the need to secure funding for further research. Continued population monitoring and habitat management are necessary well into the future, to ensure sustainable population growth of the new colony; furthermore, additional translocations would also be beneficial, as a mitigation strategy against potential inbreeding. However maintaining these efforts will not just carry benefits for future generations of the translocated chicks; they may also facilitate the implementation of translocations with other, more threatened, species on the island. Currently, opportunities to carry out such research on Corvo exist for Little Shearwaters, Manx Shearwaters *Puffinus puffinus,* and Madeiran Storm-petrel *Oceanodroma castro* (for which artificial burrows have also been built in the Reserve). Finally, it is hoped that the results from this first transfer of Cory's Shearwaters, may also inform translocation techniques for application on other islands in the archipelago, as well as to other, more threatened, species of seabird.

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Eradication and control of vertebrate invasive species in Madeira and Selvagens Archipelago: a short review

Erradicação e controlo de espécies de vertebrados invasores nos Arquipélagos da Madeira e das Selvagens: uma breve revisão

Paulo Oliveira [1] and Dília Menezes [1]

ABSTRACT - Invasive mammals are the greatest threats to island biodiversity and likely responsible for the present unfavorable conservation status of several species and many ecosystem changes in Madeira Archipelago. In response to these negative impacts of invasive species on island species and their ecosystems a systematic effort for eradicating or control them was implemented in Madeira and Selvagens archipelagos since 1985. As a result vertebrate invasive species were eradicated from 4 complex small islands and controlled locally in areas with high conservation value in Madeira mainland. This paper reviews this program assessing the approaches, successes and challenges of each project, to facilitate the conservation of islands ecosystems elsewhere in the world.

RESUMO - Mamíferos invasores são uma das maiores ameaças à biodiversidade insular e provavelmente responsáveis pelo actual estatuto de conservação desfavorável de várias espécies e muitas alterações de ecossistema do Arquipélago da Madeira. Como resposta a estes impactos negativos das espécies invasoras nas espécies e ecossistemas insulares, têm sido implementados esforços para a sua erradicação ou controlo nos Arquipélagos da Madeira e das Selvagens desde 1985. Como resultado, foram erradicadas espécies de vertebrados invasoras de 4 pequenas ilhas complexas, e controladas localmente em áreas com alto valor para a conservação na Ilha da Madeira. Este artigo faz uma revisão deste programa, avaliando as opções, sucessos e desafios de cada projecto, por forma a facilitar a conservação de ecossistemas insulares noutras partes do mundo.

Madeira archipelago is located in the Atlantic Ocean and includes two inhabited islands, Madeira and Porto Santo, and a group of non – inhabited small islands and islets, Desertas Islands (Deserta Grande, Bugio and Ilhéu Chão). Around Madeira and Porto Santo there are several other small islets; the largest of which are ilhéu da Cal and Ilhéu de Cima (Porto Santo). Selvagens archipelago is located 180 nm from Madeira and it is composed of three small non - inhabited small islands and islets, Selvagem Grande, Selvagem Pequena and Ilhéu de Fora.

Colonization of Madeira and Porto Santo occurred during the first quarter of XV century, although most authors agree that these archipelagos were known long before that time (e.g. Pereira 1956). Despite this fairly recent human presence most of the biodiversity conservation problems are related to direct or indirect human activities. High on the list of these problems are the classical

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impacts of invasive alien vertebrate species (Cabral 2005). The impact of these introduced species on the distribution and conservation status of many native species and habitats of Madeira and Selvagens Archipelago is widely documented (Silva et al. 2008, Oliveira 2010, and others).

In response to these negative impacts of invasive species on island species and their ecosystems a systematic effort for eradicating or control them was implemented in Madeira region over three decades. As a result invasive species were eradicated from many complex small islands and controlled locally in areas with high conservation value in Madeira mainland. This program has become a powerful tool to prevent extinctions and restore ecosystems in Madeira and Selvagens Archipelago. We reviewed vertebrate invasive species eradication campaigns throughout Madeira and Selvagens Archipelago. We assessed the approaches, successes and challenges of these conservation actions to facilitate the conservation of islands ecosystems elsewhere in the world.

Background information

The introduced species present in Madeira and Selvagens Archipelagos are all amongst the 100's worst vertebrate invasive species of the world and high in the list of the most widespread (Silva 2008). The number of invasive vertebrate species is clearly related to island size and human presence (Table 1). It is however surprising the fact that rats are only present in Madeira and Porto Santo Islands, despite the fact that many small islands and islets are relatively close to these islands and/or had an historical relevant human presence (e.g. Desertas Islands, Ilhéu de Cima e Ilhéu do Farol).

Madeira and Selvagens Archipelagos have a very high conservation value. The estimated number of unique endemic species of terrestrial organisms in these islands is estimated to be around 1419; 1128 in

Table 1. The presence of non native vertebrate invasive species throughout the main islands and islets of Madeira and Selvagens archipelagos in early 1980´s. The table also presents some basic information on human population, maximum altitud and size (ha) of each island or islets.

Tabela 1. *Presença de espécies de vertebrados não nativas nas principais ilhas e ilhéus dos Arquipélagos da Madeira e das Selvagens no início dos anos 1980. A tabela apresenta também alguma informação básica sobre a população humana, altitude máxima e área (ha) de cada ilha ou ilhéu.*

Island/islet	Human population	Size (ha)/ Altitude (m)	Invasive vertebrate species								Cattle
			Goat	*Pig*	*Rabbit*	*Rats (sp)*	*Mice*	*Cat*	*Ferret*	*n*	
Madeira	280.000	74175/1862	P	P	P	P	P	P	P	10	P
Porto Santo	4000	4043/517	P		P	P	P	P		5	P
Ilhéu da Cal*	(*)	140/178			P		P			2	
Ilhéu de Cima*	(*)	31/124			P		P			2	
Ilhéu de Ferro*	-	25/130			P		P			2	
Deserta Grande**	(*)	1028/479	P		P		P	P		3	
Bugio**	-	321/388	P		P		P			3	
Ilhéu Chão**	(*)	43/98					P			2	
Selvagem Gr. ***	(*)	241/163			P		P			2	
Selvagem Pq.***	-	20/49								0	

*P- Present/ n=nr. of invasive species present on each island/ *Porto Santo group of islands; ** Desertas group of islands; ***Selvagens group of islands*

Madeira Mainland, 286 in Porto Santo and its islets, 175 in Desertas Islands and 59 in Selvagens Islands (Borges et al. 2008). The eradication/control programs carried out so far are the most important instrument to protect these species, which in turn are obviously part of very important and unique ecosystems.

Eradication/control campaigns

Rodenticide in a cereal-based bait (pellets) or wax blocks was used in all but one campaign (Ilhéu de Cima) (Table 2). Wax blocks were used when it was important to have good palatable bait available during winter (Selvagem Grande and Ilhéu da Cal) or in wet conditions (Madeira), elsewhere pellets were used. Brodifacoum (0.02% and 0.05%) was consistently used because it is a very efficient second-generation anti-coagulant rodenticide (e.g. Howald et al. 2007). Bait was hand broadcasted in most campaigns following a grid of 25*25m for rabbits and 12.5*12.5 for mice. When non-target threatened species where present or the bait was exposed to severe weather conditions, bait stations were used.

The inaccessibility of Bugio island cliffs was overcome by hand broadcast of the bait from an helicopter. A specific new technique was used to accommodate the bait inside paper bags to make sure that part of the bait would be broadcasted far from the impact area and another part would stay at the place of impact. Referring to this campaign it is also noticeable the fact that goats were eradicated due to the consumption of bait. This was made possible because at the time of bait operations goat population was clustered nearby a small water supply and bait was consistently placed in the track used more often.

During the eradication campaign of Ilhéu de Cima bait was not used and most of the rabbits were caught in traps. This is a very interesting technique especially to avoid the killing of non – target species. However it is a technique that, due to the high probability of by-catch, might not work in areas were the density of breeding seabirds is very large.

All the eradication campaigns were followed by subsequent and systematic assessments by trapping, bait takes inspection and the search of other signs for at least two years (for rabbits) and three years (for mice); success was only declared when the complete removal of the target species was confirmed. When referring to the Madeira's petrel area success is measured by the decrease in bait takes and the increase in breeding success rates (Menezes et al. 2010).

Final comments

There are no doubts that the eradication/control campaigns carried out in Madeira and Selvagens Archipelagos are a very useful conservation tool. The ecosystems of most of the areas are responding very well and the conservation value of these small islands and islets was greatly improved. Using the number of plant species and invertebrates families found in study quadrats, Berthelot's pipit *Anthus bertheloti* and Gegcko *Tarentola bischoffi* densities, as indicators, Oliveira et al. (2010) presents quantitative information on habitat recovery after the Selvagem Grande project. Referring, as an example, to the average number of plant species found in the study quadrats between 2002 (pre-eradication) and 2005 (post-eradication), it is shown that this average increased from 3.8 to 7.1, becoming significantly higher in 2005 (for details refer to Oliveira et al. 2010).

For the future it is foreseen the eradication of mice *Mus musculus* from Ilhéu Chão and the creation of a new area free of invasive vertebrate species at Ponta de São Lourenço, located at the eastern far end of Madeira Island. Of major concern is the presence of goats in Deserta Grande. The eradication of this species will require a very expensive approach because it is believed that this task cannot be accomplished without the use of an helicopter during a long period of time.

It is estimated that the direct and indirect costs of the program implemented until 2012 ranged between 10 and 12 million euros.

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All the projects mentioned in this paper were implemented/are being implemented by the Serviço

Table 2. List of all the eradication/control campaigns carried in Madeira and Selvagens Archipelago, the target species, the techniques used, the outcome and the conservation value enhanced in each area.
Tabela 2. *Lista de todas as erradicações/campanhas de controlo realizadas nos Arquipélagos da Madeira e das Selvagens, espécies-alvo, técnicas utilizadas, resultados e valorização para a conservação em cada área.*

Island	Year	Type of intervention	Target species	Techniques
Madeira	1986 - ongoing	Control of an area of ca. 300 ha	Mice (*Mus Musculus*)	• Bait (waxed blocks, brodifacoum 0.02%) boxes placed strategically (n=120)
			Rats (*Rattus rattus*)	• As above
			Cats (*Felis silvestris*)	• Live traps placed strategically. Baited with fish and/or sausage. (n=25)
			Ferrets (*Mustela putorius*)	• As above
Deserta Grande	1995 - ongoing	Eradication/ control	Rabbits (*Oryctolagus cuniculus*)	• Bait (Pellets, brodifacoum 0.02%) broadcast by hand on a 25*25 m grid. A total of 15.000 tons of bait was used.
			Goats *(Capra hircus)*	• Shooting from the ground and from the sea (occasionally an helicopter was used)
Selvagem Grande	2002 - 2004	Eradication	Rabbits	• Bait (Pellets, brodifacoum 0.02%) broadcast by hand on a 25*25 m grid. After first operation grain based pellets were removed and 16.000 bait stations (waxed blocks, brodifacoum 0.02%) were placed. A total of 5000 ton of bait was used.
			Mice	• Bait (pellets, brodifacoum 0.02%) broadcast by hand on a 12.5*12.5 m grid and strategic baiting (waxed blocks and pellets, brodifacoum 0.02%)
Bugio	2007 - 2010	Eradication	Rabbits	• Year 1: Bait (Pellets, brodifacoum 0.02%) broadcast by hand on foot on half of the island. Year 2: Bait (Pellets, brodifacoum 0.02%) broadcast by hand from an helicopter on a 25*25 m. grid
			Mice	• As above
			Goats	• Bait (Pellets, brodifacoum 0.02%)placed strategically A total of 4 tons of baited was used.
Ilhéu de Cima	2010 - 2011	Eradication	Rabbits	• Shooting (20%) and trapping (80%). 25 live traps and 2600 snears were used.
Ilhéu da Cal	2012 - ongoing	Eradication	Rabbits	• Bait (Pellets, brodifacoum 0.02%) placed strategically
			Mice	• Bait (Pellets and waxe blocks, brodifacoum 0.02%) broadcast by hand on a 12.5*12.5 m grid and strategic baiting

*Success indicates that the target species is either controlled or eradicated accordingly to the “type of intervention” foreseen /**Considering Ilhéu da Cal and Ilhéu de Cima together

Outcome	Con. value enhanced
Success*	
Success	Breeding area of the endemic Madeira's Petrel. One terrestrial habitat and 14 terrestrial species present in Nature 2000 Network lists. Several other endemic species.
Success	
Success	
Success	Two terrestrial habitats and 20 terrestrial species present in Nature 2000 Network lists. Several other endemic species.
Eradication Failed/Control ongoing with success	
Success	Two terrestrial habitats and 11 terrestrial species present in Nature 2000 Network lists. Several other endemic species.
Success	
Success	Breeding area of the endemic Deserta's Petrel. Two terrestrial habitats and 16 terrestrial species present in Nature 2000 Network lists. Several other endemic species.
Success	
Success	
Success	2 terrestrial habitats and 14 terrestrial species present in Nature 2000 Network lists. Several other endemic species.*
Success	-
Ongoing	

do Parque Natural da Madeira. With the exception of Selvagem Grande, LIFE Program funds were used in all these projects (LIFE B4-3200/95/512, LIFE NAT/P/007097, LIFE 06/NAT/P/000184 and LIFE 09 NAT/PT/000041).

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Estimating density of shearwaters in the Italian sea from vessel-based surveys

Estimativa da densidade de pardelas nos mares italianos através de censos marinhos a bordo de embarcações

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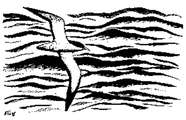

ABSTRACT - In this study we present the results of the first monitoring of Cory's and Yelkouan shearwaters (*Calonectris diomedea diomedea* and *Puffinus yelkouan*, respectively) around the Italian peninsula using vessel-based surveys. Italy hosts very large populations of these two species, holding about 10-20% of Mediterranean Cory's shearwater population, and about 25-45% of Yelkouan shearwaters (BirdLife International. 2004). From March to November 2008, we performed 15 transects per month, for a total of 11709 kilometres of transects. Both species showed a similar distribution, being most common around Sardinia and the coast of Tuscany, Campania and Apulia. Cory's shearwater were also very common west and south of Sicily, while Yelkouan shearwaters were found also in the gulf of Trieste. We also produced two Generalized Linear Models to estimate the distribution of both species around the Italian peninsula, based on environmental variables, such as distance from closer colony, distance from the coast, bathymetry, sea floor slope, Chlorophyll-a concentration, surface sea temperature. The model for Yelkouan shearwater explained 12% of deviance and performed rather well in predicting its distribution, while that of Cory's shearwater with only 5% of deviance explained, only limited areas of this species distribution.

RESUMO - Neste estudo apresentamos os resultados da primeira monitorização de cagarras e pardelas-de-yelkouan (*Calonectris diomedea diomedea* e *Puffinus yelkouan*, respectivamente) em torno da península italiana realizados em embarcações marinhas. Em Itália ocorrem populações muito grandes dessas duas espécies, com cerca de 10 a 20% da população de cagarras no Mediterrâneo, e cerca de 25 a 45% de pardelas-de-yelkouan (BirdLife International. 2004). De Março a Novembro de 2008 realizámos 15 transectos por mês, percorrendo um total de 11709 quilómetros. Ambas as espécies mostraram uma distribuição semelhante, sendo mais comuns perto da Sardenha e na costa da Toscânia, Campânia e Apúlia. As cagarras foram muito comuns a Oeste e ao Sul da Sicília enquanto que as pardelas-de-yelkouan foram também encontradas no golfo de Trieste. Em simultâneo produzimos dois Modelos Lineares Generalizados para estimar a distribuição das duas espécies em torno da península italiana, a partir de variáveis ambientais, tais como a distância à colónia, distância à costa, batimetria, declive do fundo do mar, concentração de clorofila-*a* e temperatura superficial da água do mar.

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O modelo para pardela-de-yelkouan explicou 12% da variação, sendo considerado um bom modelo para a previsão da distribuição, enquanto que o modelo de cagarra com apenas 5% de variação, limitou apenas algumas áreas da distribuição de cagarra.

In the last decades several different methods have been developed to study the distribution of seabirds, and to correlate their distribution to sea and human related features. One of the most convenient methods is to perform boat-based surveys to count seabirds. Vessel-based surveys on a modified version of Tasker's (et al. 1984) methodology, recommended by the European Seabirds at Sea Group (ESAS, Camphuysen & Garthe, 2004), can be used to obtain the density of birds, that can be also correlated to different sea features to obtain models of distribution. In 2008 LIPU has applied this method in order to obtain information regarding the distribution of seabirds and to determine the Italian marine IBAs. In this study here, we show the results of this first vessel-based survey performed in Italian waters.

METHODS

In 2008, from March to November, 139 boat-based surveys were performed all around the Italian peninsula and the major islands of Sardinia and Sicily, in order to determine the density and the distribution of pelagic birds, mainly Cory's and Yelkouan shearwaters. Most of Italian breeding pairs of the two shearwater species are found in the Tyrrenian sea and in the Channel of Sicily, where we thus concentrated our surveys. No fisherman boat have been used to realise the transect but only line ferries and sea vessels.

The transects have been performed following the methods proposed by Tasker et al. (1984) and modified by Camphuysen & Garthe (2004) including the behaviour data that allows the selections of the areas by its different uses by the species. Birds were counted between 0° and 90° (being 0° in front of the boat), and not farther than 300 m from the observer. In order to avoid double counting of individuals, the observations of flying birds were not continuous but rather single 'snapshots', performed at a frequency related to the cruise speed. In order to make the data of this study comparable to the data of the extant ESAS Database, the data of five snapshots were merged in a single statistical unit, the Poskey (Position key). This methodology allows to estimate indexes of density (observed individuals/Km²) for the different species detected in the transect and to correlate these indexes with sea features.

To each poskey we associated the following environmental features: distance from closer colony, distance from the coast, bathymetry, sea floor slope, Chlorophyll-a concentration, surface sea temperature (SST), sea condition during the transect, presence of fisherman boats or floating objects. The last three variables were recorded during the transects, together with other variables that that might have been relevant for the study, such as bird behaviour during observation or the presence of sea mammals (Camphuysen & Garthe, 2004). Average month values of chlorophyll a and SST were obtained from the NASA databases and organised in 6x6km (SST) or 12x12km (Chlorophyll-a) raster layers using ESRI ARCGIS software.

Finally, the presence of close colonies has been obtained from literature for Italy (Schenk & Torre 1986, Baccetti et al. 2005, Brichetti & Fracasso 2006, Usai et al. 2007, Baccetti et al. 2008), France (Cadiou et al. 2004, Bourgeois et al. 2008), Greece (Bourgeois et al. 2008, HOS – BirdLife Grecia pers. comm.) and for eastern Adriatic sea, Malta, Tunisia and Algeria (Bourgeois et al. 2008).

» Statistical analysis

Transect data were analysed with Generalized Linear Models (GLMs) to correlate sea distribution and density of the two shearwater species to the environmental and human related variables. For this preliminary study, we decided to use GLMs, since they offer a good and easy tool to investigate relationship between a continuous dependent variable and both continuous and class independent variables. For these analyses we included all variables and their square values, to check for non linear correlations. The best models were finally used to estimate the density and distribution of seabirds in areas not covered by sea transects.

RESULTS

We organised 15 surveys per month, from March to November 2008, for a total of 11709 kilometres of transects performed. Cory's and Yelkouan shearwaters have been the most sighted seabird species in Italian waters throughout the nine months, after the Yellow legged gull (*Larus michahellis*) (Table 1). Unfortunately, the limited amount of surveys performed in only 9 months resulted in a limited amount of deviance explained by our models, 5% for Cory's and 12% for Yelkouan shearwater. Nonetheless, these results have been used to identify those areas that, according to

Table 1. Seabirds counted during the surveys, from March to November 2008.
Tabela 1. *Aves marinhas contabilizadas durante a monitorização, de Março a Novembro de 2008.*

Species	N° individuals/100 Km								
	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Cory's shearwater (*Calonectris diomedea*)	19.1	48.85	53.77	57.84	58.66	7.85	23.46	2.28	
Yelkouan shearwater (*Puffinus yelkouan*)	3.87	29.89	16.02	17.85	34.86	2.62	0.88	4.67	1.99
Storm petrel *(Hydrobates pelagicus)*			0.18		0.2	0.04			
Gannet *(Morus bassanus)*	0.91			0.37	0.03		0.05	0.12	4.21
Cormorant (*Phalacrocorax carbo*)			0.18	0.28	0.3	0.11	0.15	0.48	4.21
European shag (*Phalacrocorax aristotelis*)			0.37	1.31	1.68	11.61	0.59		
Eleonora's falcon (*Falco eleonorae*)							0.1		
Black-headed gull *(Larus ridibundus)*	0.68			0.19	0.2	5.78	0.44	9.22	55.17
Mediterranean gull *(Larus melanocephalus)*	10.46	0.8		0.65	0.44	2.87	0.29	0.48	8.2
Yellow legged gull *(Larus machaellis)*	131.23	42.42	190.61	132.21	76.6	56.56	10.53	214.48	70.46
Audouin's gull *(Larus audoinii)*		0	24.68	0.65	0.3	0.11			
Lesser black-backed gull *(Larus fuscus)*	1.59								
Pomarine skua *(Stercorarius pomarinus)*	0.23								
Parasitic skua *(Stercorarius parasiticus)*				0.09					
Little tern *(Sterna albifrons)*			0.18						
Sandwich tern *(Stena sandvicensis)*	6.6					0.04		0.24	0.22
Gull-billed tern *(Sterna nilotica)*					0.03				
Common tern *(Sterna hirundo)*				0.09	0.5	0.04			
Caspian tern *(Stena caspia)*								0.36	
Black tern *(Chlidonias niger)*		2.57			4.22	7.96			
Whiskered tern *(Chlidonias hybridus)*					0.34				
Razorbill *(Alca torda)*									0.22

their environmental characteristics, are potentially suitable for the presence of the two species around the Italian peninsula and can be included among the candidate marine IBAs.

» Cory's shearwater

Because few Cory's shearwaters have been detected during October and November (Table 1), the models were fitted using data recorded from March to September. Cory's shearwater presence was related positively to squared bathymetry, squared Chlorophyll a and SST, while it was negatively correlated to bathymetry and squared SST (Table 2).

Its density was higher around the Tyrrenian sea, particularly around Sardinia, western Sicily, the Tuscan archipelago and Campania. Other important areas for this species were around the huge colony of Linosa (Pelagie archipelago, about 10000 pairs (Baccetti et al. 2009)), the tip of Calabria and Apulia and the area near the colony of Tremiti islands in the Adriatic sea (Fig. 1 and 2).

» Yelkouan shearwater

Yelkouan shearwater presence was positively related to squared Bathymetry and distance from coast, while it was negatively correlated to Bathymetry, squared Chlorophyll a, squared distance from coast and squared distance from closer colony (Table 3).

The GLM for Yelkouan shearwater had a good predicting value, explaining 12% of deviance. The suitable areas found included all known colonies and the area of Triest gulf, a well known foraging area for this species outside the reproductive period (Fig. 1 and 2).

DISCUSSION

In our study we tested for the first time the method of boat-based survey method suggested by Tasker et al. (1984) and modified by Camphuysen & Garthe (2004) in Italian Mediterranean sea. We obtained data about the distribution and density of Cory's and Yelkouan shearwaters during 9 months (March-November). Furthermore we calculated predicting models about the distribution of the two species around the Italian peninsula and the relevance of different sea variables for their distribution.

Cory's shearwater has been found to be the second most sighted seabird in Italian waters, after the Yellow- legged gull. The importance of Italian waters (not only territorial ones) for this species is easy to guess since Italy hosts about 10-20% of Mediterranean Cory's shearwater population. The novelty of this study based on vessel surveys is the prediction of main areas used by the species at sea. As expected, very few Cory's shearwaters have been detected in October and November when the species is its in wintering areas near the western coast of Africa. Its distribution was highly related to the presence of known colonies, such as those of Tremiti islands, Linosa, Tuscan Archipelago and the several colonies around the Sardinian Coast. The model obtained explained only 5% of the deviance,

Table 2. Model parameter estimates from GLM for the Cory's shearwater (data from survey carried out from March to September).
Tabela 2. *Estimativas dos parâmetros do MLG para a cagarra (dados da monitorização realizada de março a setembro).*

Variable	Coefficient	Standard Error	t	p
(intercept)	-8.155051	2.336773	-3.490	0.000488
Bathymetry	0.732858	0.124703	5.877	4.49e-09
Bathymetry^2	-0.109138	0.015931	-6.851	8.35e-12
Chlorophyll a^2	0.110755	0.047118	2.351	0.018788
SST	0.467918	0.225180	2.078	0.037769
SST^2	-0.010411	0.005216	-1.996	0.045974

but still predicted rather well the areas were this species could be found around the Italian Peninsula.

Yealkouan shearwater was observed less commonly than the Cory's shearwater, because although Italy hosts 25-45% of all breeding pairs of this species, this is less numerous than the Cory's shearwater by number. The distribution of this species closely resembled that of Cory's shearwater, with higher densities around Sardinia, Tuscan Archipelago, Campania and Apulian coast. The model predicted rather well its distribution (12% of deviance explained) and showed a good match with

Table 3. Model parameter estimates from GLM for the Yelkouan shearwater (data from survey carried out from March to November).
Tabela 3. *Estimativas dos parâmetros do MLG para a pardela-de-yelkouan (dados da monitorização realizada de março a novembro).*

Variable	Coefficient	Standard Error	t	p
(intercept)	-30.67888	12.48205	-2.458	0.01401
Bathymetry	0.84944	0.44701	1.900	0.05746
Bathymetry^2	-0.08985	0.04374	-2.054	0.04002
Chlorophyll a^2	-0.19566	0.07354	-2.661	0.00782
Coast distance	7.10835	2.75542	2.580	0.00992
Coast distance ^{2}	-0.41558	0.15070	-2.758	0.00584
Colony distance	-0.30603	0.10898	-2.808	0.00500

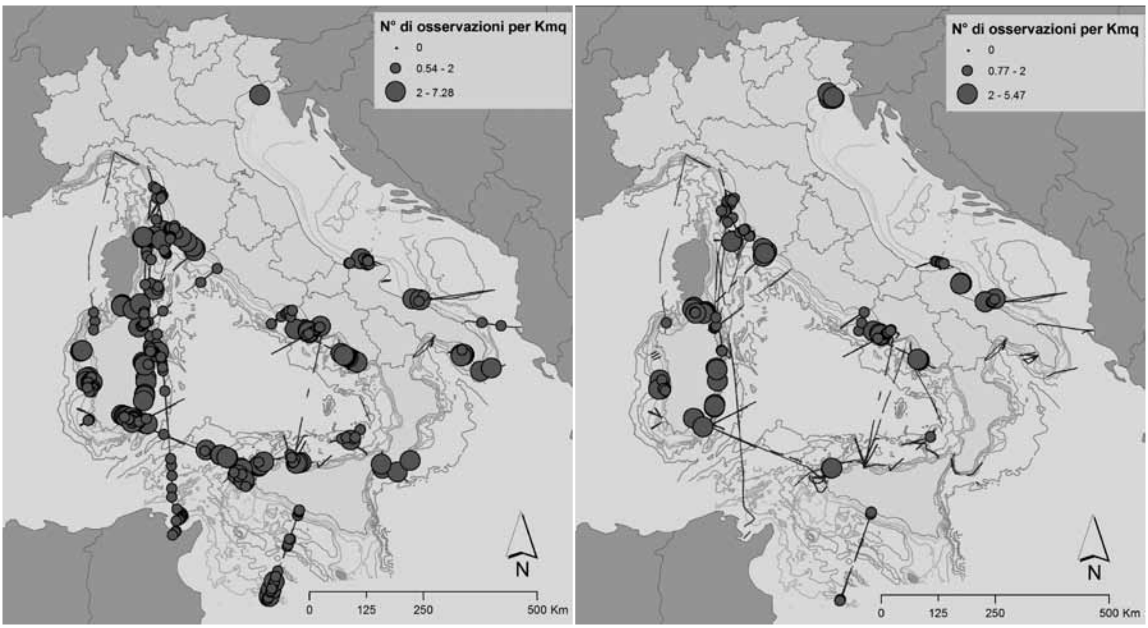

Figure 1. Cory's shearwater (left) and Yelkouan shearwater (right) densities obtained from the boat-based surveys.
Figura 1. *Densidades de cagarras (esquerda) e pardelas-de-yelkouan (direita) obtidas a partir de censos marinhos em embarcações marinhas.*

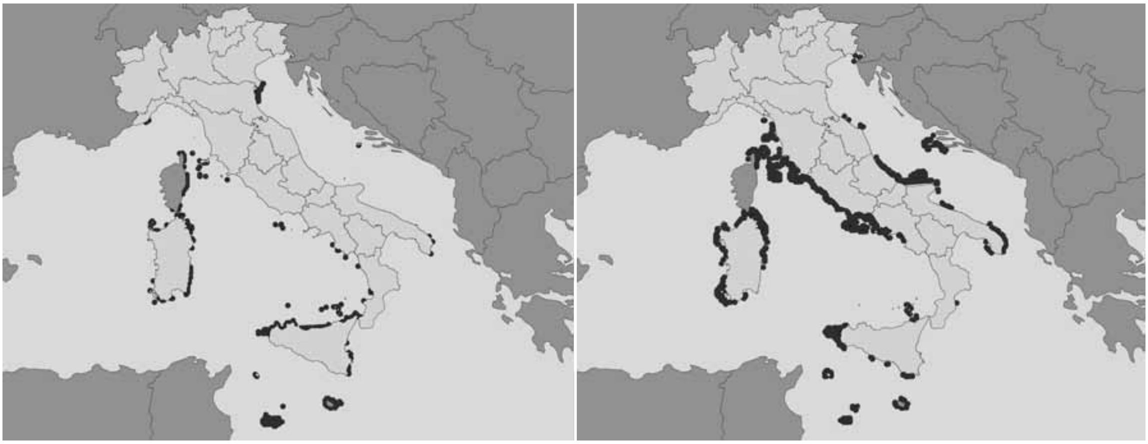

Figure 2. Estimated distribution of Cory's (left) and Yelkouan shearwater (right) as obtained by GLM. The 95% quantile (Cory's, $Q_{95\%}$ = 0.25 ind/km2; Yelkouan, $Q_{95\%}$ = 0.11 ind/km2) is showed order to represent the most used areas by the two species.

Figura 2. *Distribuição estimada de cagarras (esquerda) e pardelas-de-yelkouan (direita) obtida pelo MGL. O quantil de 95% (Cagarra, $Q_{95\%}$ = 0,25 ind/km2; pardela-de-yelkouan, $Q_{95\%}$ = 0,11 ind/km2) é mostrado para representar as áreas mais utilizadas pelas duas espécies.*

the extant data about the presence of colonies and foraging areas of this species.

This preliminary study shows that the method of vessel-based surveys is well suitable to obtain data about the distribution and the density of the two shearwater species in the Italian Mediterranean Sea. But at the same time the obtained outcomes suggest that It is necessary to carry out long term surveys (apprx. 3 years) to robustly identify main areas exploited by shearwaters and contribute to marine IBA identification.

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A Survey of Breeding Seabirds on Anguilla

Censo de Aves Marinhas Nidificantes em Anguilla

Colin Wilkinson [1], Mike Pollard [1], Geoff M Hilton 2 & Pedro Geraldes [3]

ABSTRACT - The Caribbean region supports a diverse but threatened seabird community. Conservation of Caribbean seabird populations is hampered by limited data on status and distribution of colonies. This paper reports seabird counts and estimates on Anguilla, one of the most important sites in the region. Counts were conducted from 15 May to 7 June 2007 to coincide with the peak incubation period for terns. All mainland seabird colonies and most of Anguilla's offshore cays were visited. All species were counted directly, except for a large Sooty Tern *Onychoprion fuscatus* colony on Dog Island that was estimated by sampling nest density. Sixteen seabird species were recorded. Twelve species were confirmed breeding, one was recorded as probably breeding, and three were considered as possible breeding species. The Sooty Tern colony on Dog Island was estimated at ~113,000 pairs (95% CI 85,000-143,000) and may be the largest in the Caribbean region. Three hundred Sooty Tern nests on Dog Island were monitored in the first study of nest survival in Anguilla. The predicted hatching success rate of 51% (95% CI 0.36-0.64) is within the range of results reported from other Sooty Tern nest survival studies. The data confirm the international importance of Anguilla's seabird assemblage. Based on the results, we recommend further monitoring, updating and expanding the network of protected sites on Anguilla, and comment on some habitat management issues.

RESUMO - A região das Caraíbas possui comunidades de aves marinhas diversas mas ameaçadas. Os esforços para a conservação destas comunidades são limitados pela falta de conhecimento sobre o estatuto e distribuição das colónias de aves marinhas. Este artigo descreve censos de estimativas em Anguila, um dos locais mais importantes da região. Foram efectuadas contagens entre 15 Maio e 7 Junho 2007 para coincidir com o pico da época de nidificação de Garajaus *Sterna sp.*. Foram visitadas todas as colónias da ilha principal e a maioria dos ilhéus adjacentes. Todas as espécies foram alvo de contagens absolutas, com excepção da grande colónia de Andorinha-do-mar-escura *Onychoprion fuscatus* em *Dog Island* que foi estimada por amostragem de densidades de nidificação. Foram registadas 16 espécies de aves marinhas, das quais 12 espécies nidificantes confirmadas, 1 nidificante provável e 3 nidificantes possíveis. A colónia de Andorinha-do-mar-escura em *Dog Island* foi estimada em ~113.000 casais (95% CI 85.000-143.000) e é possivelmente a maior da região caribenha. Trezentos ninhos desta espécie foram monitorizados em *Dog Island* durante o primeiro estudo de sucesso reprodutor em Anguila. A taxa de sucesso de eclosão de 51% (95% CI 0,36-0,64) encontra-se dentro dos limites dos resultados obtidos em outros estudos de sucesso reprodutor desta espécie. Os resultados confirmam a importância internacional das

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populações de aves marinhas de Anguila. Recomenda-se a continuação de acções de monitorização e actualização da rede de áreas protegidas em Anguila e referem-se algumas medidas de gestão de habitat.

Anguilla is the most northerly of the Leeward Islands, lying at 18°12' North, 63°03' West. The Anguillan archipelago comprises the mainland, the offshore cays of Anguillita, Scrub Island, Little Scrub Island, Prickly Pear East, Prickly Pear West, Dog Island and Sombrero, plus numerous other minor cays and reefs. All the islands are low-lying limestone plateaus with dense scrub cover except where cleared for development or suppressed by salt spray. Coastlines are low cliffs, bare limestone pavements, or sand beaches often backed by salt ponds (Fig. 1).

Prior to this survey, 15 seabird species have been proved to breed regularly since 1999. Additionally, Gull-billed Terns *Gelochelidon nilotica* are reported to have bred in 1964 (Bryer *et al.* 2001), at least one one Black Noddy *Anous minutus* was present on Sombrero in 1988 (Norton 1989), and Audubon's Shearwater *Puffinus Iherminieri* may have bred on Sombrero and Dog Island. Burrows attributed to the latter species were found on Dog Island in 2000, but no confirmed breeding has been reported since around 1989 (Bryer & Fisher 2000). Two pairs of Red-footed Boobies *Sula sula* bred on Prickly Pear West in 1999 and 2000 (Bryer & Fisher 2000; Holliday et al. 2007) and a pair was present beside a nest in 2004 (Collier & Brown 2004). Records of breeding 'grey' terns previously identified as Common Terns *Sterna hirundo* (Collier & Brown 2004) are now believed to relate to Roseate Terns *Sterna dougallii* (S. Holliday, *pers. comm.*). Common Terns occur in small numbers but no evidence of breeding has been found on Anguilla.

Before 1999, little definite information was available about Anguilla's seabirds, especially those on the outer cays. An Environmental Impact Assessment carried out in 1999 on behalf of Beal Aerospace included surveys of seabirds on Sombrero, and briefly examined several other offshore cays (ICF Consulting 1999). Surveys by volunteers for the Anguilla National Trust and the Royal Society for the Protection of Birds in the late 1990s indicated the importance of several of Anguilla's offshore cays for seabirds (Holliday 2000). In 2003 and 2004, surveys by Environmental Protection in the Caribbean covering most of the offshore cays recorded at least 13 species confirmed or suspected as breeding, including the first detailed estimate of the large Sooty Tern *Onychoprion fuscatus* colony on Dog Island at about 52,000 pairs (Collier & Brown 2003, 2004). These surveys showed that Anguilla holds some of the most important seabird colonies in the Lesser Antilles.

The aims of this study were to update earlier seabird surveys and establish baseline counts to help inform any future long-term monitoring. Here, we present baseline counts of all breeding seabirds on Anguilla and most of its offshore cays, and the results of the first study of nest survival in breeding Sooty Terns on Anguilla.

METHODS

» Seabird Survey

The survey was carried out from 15 May to 7 June 2007, to coincide with the expected peak incubation period for most of the tern species. On the mainland, all the salt ponds and some beaches were surveyed for Least Tern *Sterna antillarum* colonies, and undisturbed rocky coasts were surveyed for Red-billed *Phaethon aethereus* and White-tailed Tropicbirds *P. lepturus*. Boat landings were made on Prickly Pear West on 18 May, Dog Island from 20-24 May and again on 4 June, Scrub Island from 26-27 May, and Anguillita and Prickly Pear East on 2 June. In every case, at least one complete circuit of the coastline was made on foot. Little Scrub Island was surveyed on 27 May from the closest part of Scrub Island and by a close circuit by boat. We were not able to visit Sombrero.

Except for the Sooty Terns on Dog Island, direct counts of all adults, fledged juveniles, chicks, eggs or nests were made as appropriate to the species,

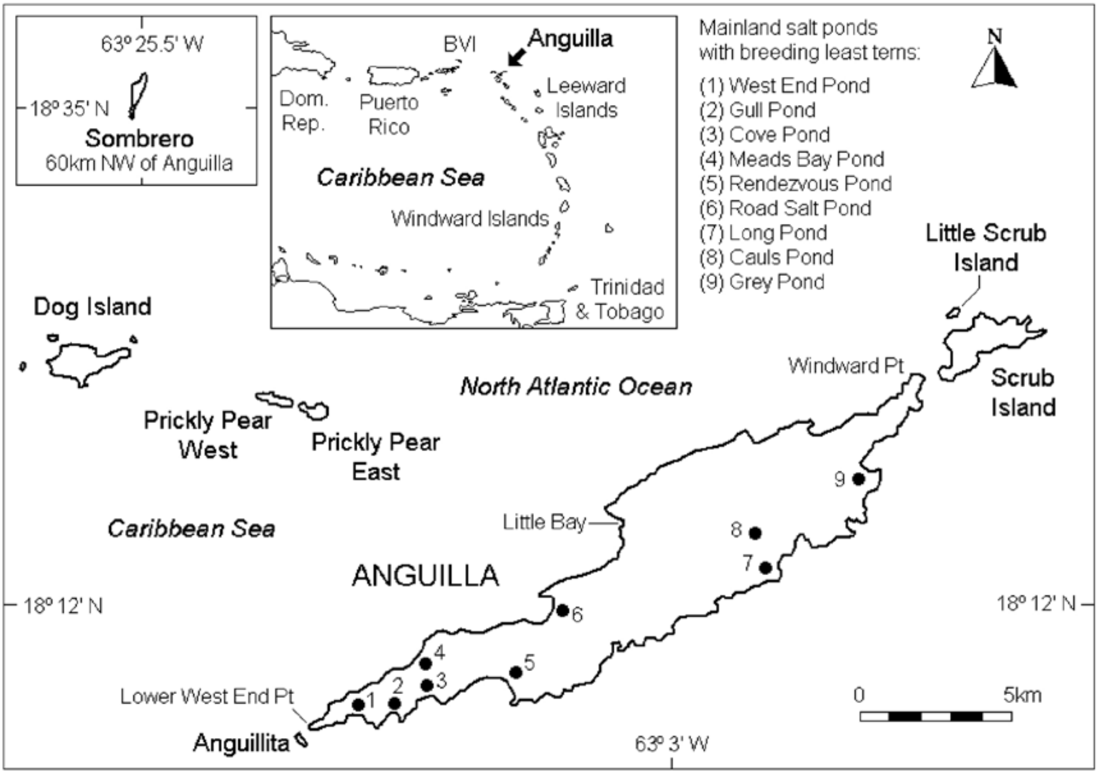

Figure 1. Location of Anguilla and of study sites.
Figura 1. *Localização de Anguila e locais de estudo.*

ground conditions and breeding stage, which enabled estimates of the numbers of apparently occupied nests (AON) to be made (Bibby *et al.* 1992). In some cases where nests were hard to count, mainly Laughing Gulls *Larus atricilla* and Bridled Terns *Sterna anaethetus*, it was feasible only to count the number of adults present and estimate the number of pairs represented. In most cases, this was likely to under-estimate the true number as some adults will have been away from the colony during the counts.

Tropicbird nest sites were located by watching for visiting adults, and searching carefully for nest cavities in suitable habitat. Possible breeding was recorded if adult(s) demonstrated prolonged interest in an area with suitable nesting habitat but did not land, or if adult(s) were in a potential nest site, but no egg or chick was visible. Confirmed breeding was recorded at sites with an egg, chick or immature, or where evidence of recent nest site occupation (feathers, droppings or dead chick) was found.

» Dog Island Sooty Tern Census

The large colony of Sooty Terns on Dog Island was censused by sampling nest density and mapping the colony area (Fig. 2). On 21-22 May 2007, 28 circular sample plots of 50m^2 were surveyed. The aim was to establish plots as systematically as possible, ~50m apart, in lines from the southern edge of the colony as far across it as possible, ideally to the opposite, northern edge. Parallel lines of plots were separated by at least 200m. In practice, impenetrable thorn scrub often forced modifications to the sample pattern. In each plot, intact eggs, predated eggshells, and incubating adults were counted. These data gave estimates of the numbers of pairs, and hence the pair-density (pairs/

m^2) for each plot. The perimeter of the colony was recorded using a Global Positioning System, and the area subsequently calculated using ArcGIS 9.1 software. In addition to the 28 new plots, we relocated the 4 circular 100m^2 plots established by Collier & Brown in 2004, and carried out repeat counts. Collier & Brown (2004) established their plots by random sample. Because of the differences in approach and the fact that three of these plots were found to lie outside the current limits of the colony, data from these 4 plots was not included to help calculate our population estimate or to provide a repeat estimate using just those 4 plots.

There were significant differences in habitat and pair-density between different parts of the colony. Plots were not distributed in a perfectly systematic way across the colony, and some areas were over or under-represented, so we split the data into different sectors of the colony and repeated the extrapolation exercise. The colony was divided into 3 sectors: west (containing 11 plots), centre (8 plots) and east (9 plots), based on the apparent differences in habitat and pair density. The east sector was covered mainly in low (<0.5 m high) non-thorny scrub while much of the west sector comprised particularly dense, tall thorn scrub up to 2 m high. These habitat differences may have been reflected by the differences in observed pair densities across the 3 sectors (Table 2).

We used the pair-density estimates, combined with the overall area of the colony, to estimate the total number of breeding pairs as follows. The observed list of plot pair-densities was bootstrapped 999 times to give 999 average pair-density values. The median of these values was taken as average pair-density for the sector, with 95% confidence limits also calculated from the bootstrapping results. The bootstrap was performed separately on the

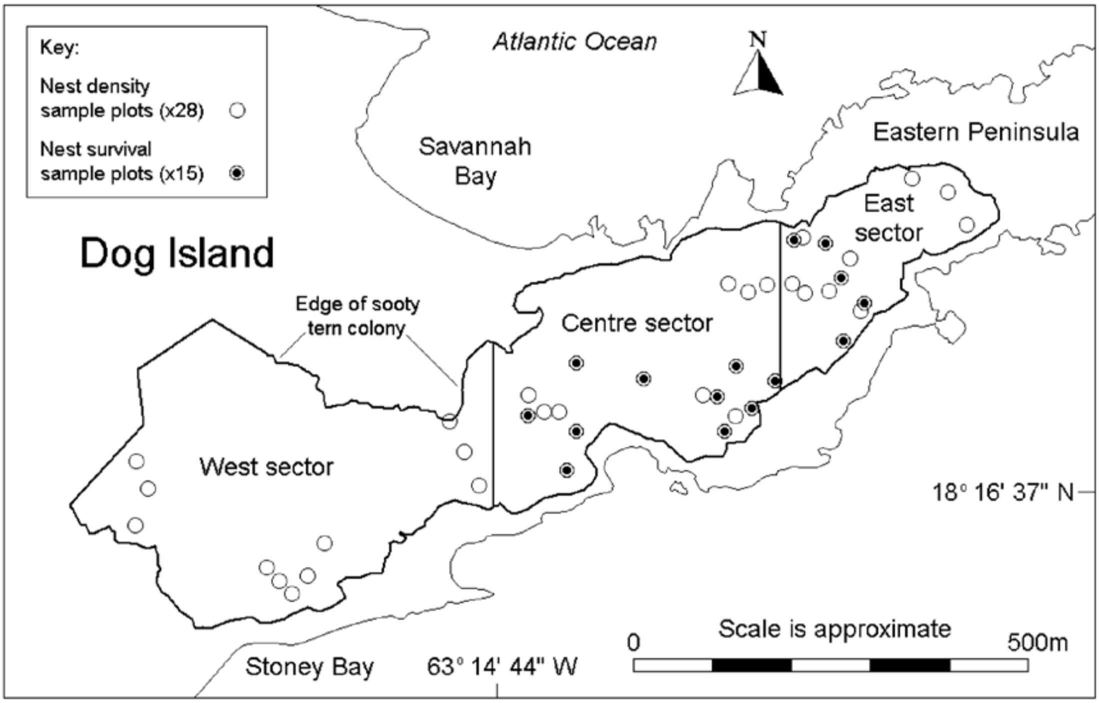

Figure 2. The Sooty Tern colony on Dog Island, showing nest density sample plots and sectors used in analysis, and locations of nest survival sample plots. Area of sample plots not to scale.

Figura 2. *Colónia de andorinha-do-mar-escura em Dog Island com pontos de amostragem de densidades de nidificação e sectores utilizados para análise e localização de pontos de amostragem de sucesso reprodutor. A área de pontos de amostragem não se encontra à escala.*

data from each sector, and the extrapolations were performed separately using the respective areas of each sector. The median pair-density, and associated 95% confidence limits were extrapolated to the total area of the colony, to give an overall estimate of the number of nesting pairs for each sector as follows:

Total pairs = (bootstrapped median density x (sector area – area of the plots)) + number of pairs in plots

The sector estimates were then added together.

The three-sector analysis showed that the east and centre sectors had very similar, and statistically indistinguishable, pair-densities. The west sector was significantly different. We therefore ran a final analysis, combining the east and centre sectors, but keeping the west sector separate. The bootstrap and extrapolation was repeated for the combined centre/east sector and for the west sector.

Since all sample counts were completed before any eggs hatched, there will have been no under-estimate due to hatched chicks leaving the sample plots. However, as the sample counts took place at an advanced stage of incubation, there was scope for significant under-estimate due to pairs failing at incubation stage before the census took place – unless pairs re-laid before the count dates, most of these would not have been recorded by the observers. The Sooty Tern nest survival study helps us judge the potential scale of under-estimation; this is dealt with in the results.

» Sooty Tern Nest Survival

Three hundred nests were marked in the Dog Island colony on 8-9 May 2007, divided among 15 plots of 20 nests each. The plots were widely separated and chosen non-randomly to represent a range of typical nesting habitats and situations within the colony, including in tall thorn scrub, low non-thorny scrub, and at varying distances from the colony edge. Each plot was marked visually and with a Global Positioning System, and at each location, the nearest 20 nests found containing eggs on 8-9 May were marked. All nests contained single eggs, except for one that contained 2 eggs. Nest contents were checked on 15-16 May, 22-23 May and 4 June 2007. At each visit, data were recorded on nest contents. No chicks were seen until 4 June, so empty nests recorded on 15-16 May and 22-23 May were assumed to have failed rather than hatched.

To analyse nest survival during incubation, we used data from the first two time-intervals (8-9 May to 15-16 May, and 15-16 May to 22-23 May). This period covers approximately half of the total incubation period for the species. Based on a modified Mayfield analysis (Mayfield 1975), we coded each nest according to the outcome (1= failed, 0 = successful) and the number of 'exposure days' (days on which the nest was known or inferred to have been still active). If a nest failed during a particular time interval, it was assumed to have failed at the mid-point of this period and the number of exposure days was calculated accordingly. Daily nest survival was then estimated using a binomial model with a logit link function. To account for the potential non-independence of nests within the same plot, we used a Generalised Linear Mixed Model (GLMM, Proc GLIMMIX in SAS v.9) to model nest survival, with 'plot' declared as a random factor.

More sophisticated approaches to the statistical analysis of nest survival are available (Rotella et al. 2004, Grant et al. 2005), but with only two time intervals for all birds, their use was not justified because of the simplicity of this data set.

RESULTS

» Seabird Survey

During the survey period (Table 1), 12 species of seabirds were confirmed breeding in Anguilla (not including Sombrero). In addition, White-tailed Tropicbirds were recorded as a probable breeding species. Roseate Terns, Common Terns and a single Red-footed Booby were observed near suitable nesting habitat and were counted as possible breeding species.

Sooty Terns were overwhelmingly the most numerous species, with an estimated 113,000 pairs (95% CL = 85,000-143,000 pairs) in the main colony on Dog Island. Very small numbers of Sooty Terns were also seen on Scrub and Little Scrub Islands and the Prickly Pears, though breeding at these sites was not confirmed. Bridled Terns and Brown Noddies *Anous stolidus* were the most widespread species; breeding was proved or suspected on every island visited including the mainland, where several breeding pairs of these two species at Lower West End Point constituted the first confirmed mainland

Table 1. Estimated number of pairs of breeding seabirds in Anguilla, May-June 2007. | ***Tabela 1.*** *Estimativa do número de casais de aves marinhas nidificantes em Anguilla, Maio-Junho 2007.*

Species	Mainland	Anguillita	Scrub Island	Little Scrub	Prickly Pear East	Prickly Pear West	Dog Island	Estimated total pairs	Breeding status
White-tailed Tropicbird *Phaethon lepturus*	2-3							2-3	Probable
Red-billed Tropicbird *Phaethon aethereus*	11	1	4		1	11	15	43	Confirmed
Masked Booby *Sula dactylatra*							42	42	Confirmed
Brown Booby *Sula leucogaster*				20	40	700	600	1,360	Confirmed
Red-footed Booby *Sula sula*						1		1	Possible
Brown Pelican *Pelecanus occidentalis*						29		29	Confirmed
Magnificent Frigatebird *Fregata magnificens*							310	310	Confirmed
Laughing Gull *Larus atricilla*		145	200	38	875	39	365	1,662	Confirmed
Royal Tern *Sterna maxima*		12						12	Confirmed
Sandwich Tern *Sterna maxima*		7						7	Confirmed
Roseate Tern *Sterna dougallii*	175		15					190	Possible
Common Tern *Sterna hirundo*			2					2	Possible
Least Tern *Sterna antillarum*	268		20		29		3	320	Confirmed
Bridled Tern *Sterna anaethetus*	4	45	2	7	24	10	46	138	Confirmed
Sooty Tern *Onychoprion fuscatus*			1	2	6	3	113,000	113,000	Confirmed
Brown Noddy *Anous stolidus*	9	8	2	53	13	9	191	285	Confirmed

breeding records. Laughing Gulls and Red-billed Tropicbirds were also widespread. By contrast, Masked Boobies *Sula dactylatra* and Magnificent Frigatebirds *Fregata magnificens* were confirmed breeding only in localised colonies on Dog Island in 2007, although Masked Boobies have also previously been recorded on Sombrero (ICF Consulting 1999, Holliday 2000). The only colony of Brown Pelicans *Pelecanus occidentalis* was on Prickly Pear West, and the only Royal Terns *Sterna maxima* and Sandwich Terns *Sterna sandvicensis* found breeding were on Anguillita. However, mating pairs of both these tern species were also seen on Scrub Island. At one mainland site, Little Bay, 5 White-tailed Tropicbirds were seen investigating potentially suitable nesting cliffs. Although more than 300 Roseate Terns were seen around Scrub Island and nearby parts of the mainland and a few pairs were seen mating, no breeding was confirmed. On Scrub Island 4 Common Terns were seen, but they showed no sign of breeding activity.

» Dog Island Sooty Tern Census

Pair-density was high in the centre and east sectors of the colony, but much lower in the larger, western sector (Table 2). Estimates of the total number of pairs present were broadly similar for the three different treatments of the sector data (Table 3). Combining data for all sectors into a single bootstrap/extrapolation resulted in the highest estimate, because the west sector, which has a low pair density, was somewhat under-represented in the sample of plots. However, the slightly lower estimate for the two-sector data treatment was the preferred one, since it captures the major source of spatial variation – that between the west and the other two sectors – and gives a relatively low coefficient of variation.

The census took place on 21-22 May, prior to any hatching. The great majority of eggs hatched by the last visit on 4 June. Hence, median hatch dates likely lay between ~25 May and 1 June. Consequently, the census took place on average ~70-90% (=19-26 days) of the way through incubation, and therefore under-estimates the true number of pairs that nested, due to nest failures prior to the census. Nest failure rate was estimated at 0.0228 per day (see below) and therefore we can predict that ~35-45% of nests might have failed prior to the census. The observers would have recorded very

Table 2. Pair density and sector areas for the Sooty Tern colony on Dog Island.
Tabela 2. *Densidade de casais e áreas de cada sector da colónia de andorinha-do-mar-escura em Dog Island.*

Colony sector	Pair density (pairs m^{-2}) (95% CI)[1]	Sector area (ha)	Extrapolated number of pairs (95% CI)
West Sector	0.21 (0.10-0.35)	13.9	29,800 (13,700-47,900)
Centre Sector	0.46 (0.21-0.71)	10.1	46,900 (21,400-72,000
East Sector	0.50 (0.29-0.74)	4.8	24,500 (14,300-35,700)
Colony Total	0.42 (0.30-0.55)	28.9	101,300 (69,900-133,600)

[1] Median and 95% confidence limits calculated by bootstrapping. | [1] *Mediana e intervalos de confiança 95% calculados por* bootstrapping.

Table 3. Sooty Tern colony size on Dog Island based on different data treatments.
Tabela 3. *Dimensão da colónia de andorinha-do-mar-escura em Dog Island com base em diferentes tratamentos de dados.*

Data treatment	Estimated nesting pairs (95% CI)
All sectors combined	122,100 (85,500-158,800)
Three sectors (centre, west, east)	101,300 (69,900-133,600)
Two sectors (centre + east, west)	113,600 (85,000-143,300)

few adults that had already failed in their breeding attempt. Therefore, it is likely that the true nesting population was much higher than the census figures indicate, though it is difficult to estimate this precisely without information on temporal variation in nest failure rate.

» Sooty tern Nest Survival

Allowing for the 3 lost markers up to check 2 on 22-23 May, the original sample size amounted to 298 eggs. Twelve percent of the eggs disappeared by check 1, seven days later. A further 18% disappeared in the next seven days from check 1 to check 2. No eggs had hatched by check 2, on 22-23 May. At check 3, 52% of the eggs present on the previous visit were confirmed to have hatched, while only 10% were still present and intact and 1% had probably been predated since 22-23 May. There was no trace of the remaining 37% of eggs or chicks (Table 4).

Thus, hatching was very synchronous. Because all egg disappearances prior to the 22-23 May visit occurred before hatching, they can all be ascribed to nest failure during incubation. Conversely, because the majority of hatching took place between the 22-23 May and the 4 June visit, egg survival during that interval cannot be determined. Furthermore, because chicks become mobile soon after hatching, and therefore difficult to detect, it is not possible to estimate nest survival over this interval with any confidence. The fate of 89 eggs was undetermined, and it is not possible to determine whether they failed at egg stage, chick stage, or produced live chicks that had moved from the marked nest sites.

The Generalised Linear Mixed Model of nest survival, using failures/exposure days as the binomial response variable, and plot as a random effect, indicated that overall daily nest failure rate was 0.0228 (95% CI 0.015–0.034). This gives predicted hatching success of 51% (95% CI 0.36-0.64) for an incubation period of 29 days (Ashmole 1963, del Hoyo et al. 1996, Higgins & Davies 1996), if the assumption of constant failure rate throughout incubation is correct.

A large proportion of the variance in nest survival occurred at the plot level (Covariance Parameter for Plot random effect =0.50 - SE =0.26), indicating considerable spatial variation in nest survival across the colony.

Inclusion of a two-level fixed explanatory factor ('location'), indicating whether the plot was in the 'core' or 'periphery' of the colony did not improve the model. The 'location' variable was not significant ($F_{1\ 13,1}$ =0.10 - P =0.76). There were no striking spatial patterns of nest success variation across the study area, with successful plots occurring close to relatively unsuccessful plots.

DISCUSSION

Censusing tropical seabirds is fraught with difficulty. Large inter-annual variations in timing of breeding are frequently exhibited in many populations of tropical seabirds (Schreiber & Schreiber 1986). Therefore, any comparison of just two points in time cannot reliably be used to infer trends. However, our data clearly confirm

Table 4. Sooty Tern nest survival study on Dog Island. | ***Tabela 4.*** *Estudo do sucesso reprodutor de andorinha-do-mar-escura em Dog Island.*

Nest status	Visit 1 8 May	Visit 2 15-16 May	Visit 3 22-23 May	Visit 4 4 June
Intact eggs	301	262	207	20
Empty nests	-	27	87	156
Predated eggshells	-	9	2	2
Hatched eggshells	-	0	0	18
Live chicks	-	0	0	89
Dead chicks	-	0	0	1
Markers lost	-	3	3	15

the international importance of Anguilla's seabird populations. This conclusion is in spite of the lack of any data for Sombrero in 2007, and the fact that the peak egg-laying season for Brown and Masked Boobies and Magnificent Frigatebirds probably occurred over the previous autumn and winter.

Compared to results reported by Bryer & Fisher (2000), Holliday (2000) and Collier & Brown (2004), we recorded substantially higher numbers of Brown Pelicans, Magnificent Frigatebirds, Red-billed Tropicbirds and Sooty Terns. For the latter two species, some of the increase is attributable to enhanced survey effort. Increased time spent searching suitable habitat is likely to give truer estimates of tropicbird populations since these species are not strictly colonial breeders, have extended breeding seasons, and the nest sites are often well hidden (Lee & Walsh-McGehee 2000, McGowan et al. 2006).

Estimates of the Sooty Tern population on Dog Island have increased significantly with almost every new survey since the 1990s. Sooty Terns nest in dense, highly synchronous colonies, so are relatively easy to census compared to some species. Sooty Terns are among the most numerous seabirds in the world, with an estimated world population upwards of 25 million pairs (del Hoyo et al. 1996). Some nest in spectacularly large colonies – colonies of several hundred thousand pairs or more are known in several Indian Ocean and Pacific Ocean locations (del Hoyo et al. 1996, Feare et al. 2007). However, in the Atlantic and Caribbean region, the Dog Island population assumes considerable importance. The Sooty Tern colony on Ascension Island comprises ~200,000 pairs (Sanders 2006), but we are aware of no other similarly large colonies in the tropical and sub-tropical Atlantic (Croxall et al. 1984). None of the Caribbean colonies are thought to reach 100,000 pairs, though there are a few with >50,000 pairs, and several large ones have not been quantified (Schreiber & Lee 2000). Dog Island may be the largest known Sooty Tern colony in the Caribbean, and the second largest in the Atlantic-Caribbean region. Nevertheless, even this study may have under-estimated the true number of pairs at Dog Island by up to ~35-45%, simply by conducting the census at an advanced stage of incubation.

Counts of White-tailed Tropicbirds, Brown, Masked and Red-footed Boobies, and Bridled Terns are broadly consistent with previous surveys (allowing for the lack of data for Sombrero in 2007). Counts of Laughing Gulls should be interpreted with care as pair estimates were based only on flush counts of flying adults. We found fewer Brown Noddies on Little Scrub and Prickly Pear East than previously recorded; elsewhere the results are broadly consistent with previous surveys.

Least Tern numbers are broadly consistent with previous surveys, but the total includes several new colonies including on Prickly Pear East. In contrast, we recorded smaller numbers at some of the mainland colonies than previous surveys did. A large amount of tourism-related development has occurred on the Anguilla mainland since 2000, affecting several ponds favoured by Least Terns. In many cases, the area of open marginal salt flats they nest on has been reduced by rubble infill. Several colonies are now very close to roads and built development, and may in future be more vulnerable to unintentional disturbance and associated predation. Several more currently undisturbed sites are earmarked for large-scale development. Undisturbed breeding habitat for Least Terns is increasingly rare on Anguilla and if development continues at the current rate, future surveys may well record significant declines in both numbers and reproductive output of the species.

The absence of any breeding Roseate Terns in this survey was surprising, but highlights the well-documented tendency of this species to shift or abandon breeding sites abruptly (Nisbet 1989, Shealer & Burger 1992).

» The importance of Anguilla's seabird colonies

To gauge the importance of Anguilla's seabird populations, the results can be compared to the global and Caribbean population estimates (Table 5). By convention, any site that supports 1% or more of the relevant biogeographical population is classed as 'important' according to global prioritisation schemes such as Important Bird Areas (IBA's) and the Ramsar convention. Anguilla does not surpass the global 1% thresholds for any species although the populations of Brown Boobies, Least Terns and Sooty Terns are of some note in this

Table 5. The importance of Anguilla's seabird populations in a regional and global context. | ***Tabela 5.*** *A importância das populações de aves marinhas de Anguila num contexto global e regional.*

0o	Anguillan population from this study (individuals)	Sombrero population (most recent data)[1]	Caribbean Population[2]	Anguilla as % of Caribbean population	Global population	Anguilla as % of global population
White-tailed Tropicbird *Phaethon lepturus*	5		6,000	0.08	50,000[3]	0.01
Red-billed Tropicbird *Phaethon aethereus*	86		4,300	2.00	20,000[3]	0.43
Masked Booby *Sula dactylatra*	84	40	1,200	10.33	200,000[3]	0.06
Brown Booby *Sula leucogaster*	2,720	772	13,300	26.26	300,000[3]	1.16
Red-footed Booby *Sula sula*	1		18,200	0.006	1,000,000[3]	~0.00
Brown Pelican *Pelecanus occidentalis*	58		3,000	1.93	250,000[4]	0.02
Magnificent frigatebird *Fregata magnificens*	620		9,600	6.46	300,000[3]	0.21
Laughing gull *Larus atricilla*	3,324	6	15,000	22.20	800,000[5]	0.42
Royal Tern *Sterna maxima*	24		1,250	1.92	139,000[6]	0.02
Sandwich Tern *Sterna maxima*	14		5,100	0.27	200,000[5]	0.01
Roseate Tern *Sterna dougallii*	382		10,000	3.82	100,000[5]	0.38
Common Tern *Sterna hirundo*	5		150	3.33	750,000[5]	~0.00
Least Tern *Sterna antillarum*	640		4,500	14.22	120,000*[5]	0.53
Bridled Tern *Sterna anaethetus*	276	368	12,000	5.37	400,000[5]	0.16
Sooty Tern *Onychoprion fuscatus*	226,000 (95% CI 170,000-286,000)	332	500,000	45.27 (34-57%)	50,000,000[5]	0.45 (0,34-0,57%)
Brown Noddy *Anous stolidus*	570	654	28,000	4.37	800,000[5]	0.15

1: ICF Consulting (1999), 2: Schreiber & Lee (2000), 3: del Hoyo et al. (1992), 4: Rose & Scott (1997), 5: del Hoyo et al. (1996), 6: Wetlands International (2006).
*Least tern population given here is for North American and Caribbean populations only.

context. Anguilla does hold more than 1% of the Caribbean population of 11 seabird species found breeding in 2007, plus two more not confirmed breeding in our surveys. As Sombrero could not be visited during 2007, table 5 incorporates the most recently available comprehensive data on the island (ICF Consulting 1999).

Under the Ramsar Convention, a wetland should be considered internationally important if it regularly supports 20,000 or more water birds (criterion 5) or if it regularly supports one percent or more of the individuals in a population of one species or subspecies of water bird (criterion 6). Ramsar sites in Anguilla were most recently assessed in 2005 by the UK Overseas Territories Conservation Forum (Pienkowski 2005). Sombrero, Dog Island, the Prickly Pears, Scrub and Little Scrub Islands, and Cauls Pond, Cove Pond, Savannah Pond and Road Salt Pond on the mainland were proposed for a range of criteria. This assessment should be reviewed in light of the results presented here. Dog Island should be recognised as qualifying under criterion 5, and Anguillita should be added to the suite of proposed Ramsar sites on criterion 6, for Laughing Gulls and Royal Terns. Finally, despite having first being reviewed against Ramsar criteria as long ago as 1990 (Pritchard 1990), no sites on Anguilla have been formally designated. Given the pace and scale of development on Anguilla, the need to protect the most important seabird sites is more urgent than ever.

» Sooty Tern Nest Survival

We estimate that 51% of Sooty Tern eggs hatched on Dog Island in 2007. Therefore, of an estimated number of 113,000 nesting pairs, ~55,400 nests failed at the incubation stage. Sooty Tern hatching (and fledging) success appears to be highly variable across years, in different parts of colonies and between colonies. A success rate of 51% is not remarkable compared to other reported values, which range from 7% on Little Tobago (Morris 1984) to 75% on Bird Island in the Seychelles (Feare 1976). Where mammalian predators are present, success tends to be low. Two authors reported substantial differences in hatching success between core and periphery, which were not detected on Dog Island (Feare 1976, Hazin & Macedo 2006). Where this occurred in other sites, it was because predators were able to access the colony periphery, but not the core. The lack of such a pattern at Dog Island may suggest that predation is not the main cause of nest failure, or that the density and/or habitat of the colony allows predators to reach the core to the same extent as the periphery.

Several introduced or native species present on Dog Island could have contributed to these losses. Some eggs may be lost to trampling by goats, of which 200-250 were present in May 2007. The introduced Black Rats *Rattus rattus* predate eggs, as confirmed by analysis of rat stomach contents (Varnham 2007). There are several species of land crabs present – Hazin and Macedo (2006) identified a land crab *Geocarcinus lagostoma* as being the chief predator of Sooty Tern eggs on Ascension Island. Finally, on Little Scrub, the endemic ground lizard *Ameiva corax* is known to predate Bridled Tern eggs (Hodge et al. 2003). Males of the ground lizard *Ameiva plei plei*, which is common on Dog Island, grow to a larger size than *A. corax* and may predate some Sooty Tern eggs. One male was seen tapping an intact Sooty Tern egg with its nose during the survey.

As a long-lived species with low productivity rates and delayed maturity, Sooty Tern demography is relatively insensitive to moderately low breeding success or periodic poor years. There is little information on the long-term levels of reproductive success required for population maintenance, or on long-term trends at Dog Island. The species is notable for persisting at islands following the invasion of mammalian predators. For example, a large Sooty Tern population persisted on Ascension Island after all other seabird populations had been extirpated from the main island by cats and rats (Ratcliffe et al. 1999, Hughes & Wearn 2005). Similarly, in the Chagos Archipelago, Sooty Terns persist on rat-infested islands where very few other seabirds remain (Symens 1996). This may arise because of the synchronous nesting habit, which results in predator swamping. However, the ability to avoid acute declines and extirpation does not imply that the species is entirely insensitive to introduced predators. There is considerable potential for large, synchronous colonies to decline relatively slowly due to chronic nest failure and for this to go unnoticed.

Recent population modelling for *Pterodroma* petrel species – which have similar demography to Sooty Terns – indicates that very large populations can undergo declines lasting several centuries as a result of high predation rates, without actually being extirpated (Brooke et al. 2010).

» Conservation Management Issues

Prolonged browsing pressure from goats probably has led to the thorn and cactus-dominated vegetation on Dog Island. In the east sector of the colony, Sooty Terns nested in low, non-thorny scrub and much of it showed stunted growth due to browsing. Trampling by goats may also be responsible for at least some of the lost Sooty Tern eggs.

Browsing pressure may help maintain open patches within the scrub that allow the terns to land and take off near their nest sites.

Black Rats were present on Dog Island, Scrub Island and probably on Prickly Pear East. Rats are likely to reduce breeding success of several seabird species on the islands; there is quantitative evidence from other islands that rats can substantially reduce breeding success of Brown Noddies, Black Noddies, Red-tailed Tropicbirds, Audubon's Shearwaters, and Bridled, Roseate and Sooty Terns (Towns et al. 2006). With the support of the data gathered during this survey, in 2012, Fauna & Flora International (FFI), the Anguilla National Trust, the Government of Anguilla, the Royal Society for the Protection of Birds (RSPB), and the landowner, Anguilla Development Company have eradicated alien invasive black rats from Dog Island and two neighbouring cays. The project team is now monitoring the effects of removing the rats, as well as establishing fences to exclude feral goats from the most sensitive areas.

Generally, human disturbance on the offshore cays currently seems to be low. Where landings do occur (notably on Dog Island, Scrub Island and Prickly Pear East) tourist activity seems largely to be limited to the main beaches and seems unlikely to have significant impacts on breeding seabirds. However, significant disturbance could arise from any landing, if it was at a sensitive time of year and visitors were not aware of the impact of their behaviour. Publicity and education initiatives on the mainland, and advisory notices placed on the popular landing beaches may help reduce this risk.

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Seasonal abundance and population trends of waterbirds at Caldeira de Tróia, Sado estuary

Abundância sazonal e tendências populacionais de aves aquáticas na Caldeira de Tróia, estuário do Sado

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ABSTRACT - The Sado estuary is the second largest estuary in Portugal and it has been classified as an internationally important wetland under the Ramsar Convention. Caldeira de Tróia is a small tidal lagoon connected to the main estuary and, thus, being under the influence of the same tidal regime. Between 2004 and 2012, monthly surveys were carried out during low tide at Caldeira de Tróia to assess the seasonal abundance of waterbirds. Wintering population trends were determined for the most common species and compared with those from the whole estuary. The results show that although comprising less than 2% of the total estuary mudflat area, the Caldeira de Tróia holds proportionally large numbers of several species such as the Dunlin *Calidris alpina*, Sanderling *Calidris alba*, Ringed plover *Charadrius hiaticula*, Kentish plover *Charadrius alexandrinus*, Greenshank *Tringa nebularia* and Turnstone *Arenaria interpres*. Waterbird abundance peaks during the autumn/winter months, decreasing significantly in spring and early summer. The tidal flats of the Caldeira de Tróia hold low number of waders during migratory periods. Exceptions are Whimbrels *Numenius phaeopus* which are more abundant during spring migration (April-May) and Kentish plovers, peaking in numbers during the autumn migration (August-October). The few existing data from surveys conducted at the Sado estuary outside the winter period seems to generally support the phenological pattern described for waders at the Caldeira de Tróia. More research is clearly needed but these data suggest that the Sado estuary has reduced importance as a stopover site mainly during spring migration, especially when compared to the neighbouring Tagus estuary. The wintering populations of Ringed plover, Kentish plover, Dunlin and Turnstone showed a negative trend at the Caldeira de Tróia during the study period. However, these trends were not consistent with those recorded for the entire estuary in the same period.

RESUMO - O estuário do Sado é o segundo maior estuário de Portugal e está classificado como zona húmida de importância internacional sob a Convenção de Ramsar. A Caldeira de Tróia é uma laguna do estuário que forma uma reentrância na península de Tróia, estando ligado ao estuário e portanto sob a influência das marés. Neste último local realizaram-se contagens mensais de baixa-mar entre 2004 e 2012 com o objectivo de descrever as abundâncias sazonais da comunidade de aves aquáticas, descrever tendências populacionais para as espécies mais comuns e compará-las com as tendências globais registadas em todo o estuário. Os resultados deste estudo

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demostram que, apesar da sua reduzida dimensão, a Caldeira de Tróia desempenha um papel importante como área de alimentação para um número elevado de espécies presentes no estuário do Sado, nomeadamente o Pilrito-comum *Calidris alpina*, o Pilrito-d'areia *Calidris alba*, o Borrelho-de-coleira-interrompida *Charadrius alexandrinus*, o Borrelho-grande-de-coleira *Charadrius hiaticula*, o Perna-verde *Tringa nebularia* e a Rola-do-mar *Arenaria interpres*. O número máximo de aves aquáticas ocorre nos meses de Outono/Inverno, decrescendo significativamente nos meses de Primavera e início do Verão. A Caldeira de Tróia concentra pequenos números de aves limícolas durante os meses de migração. Apenas o Maçarico-galego *Numenius phaeopus* e o Borrelho-de--coleira-interrompida apresentam claros picos de abundância durante os períodos de migração pré-nupcial (Abril-Maio) e pós-nupcial (Agosto-Outubro), respectivamente. Os escassos censos efectuados no estuário do Sado fora do período de inverno parecem suportar na generalidade o padrão fenológico registado para as aves limícolas na Caldeira de Tróia. Embora exista claramente a necessidade de comprovar estes resultados com censos adicionais, os dados aqui apresentados sugerem que a importância do estuário do Sado como zona de *stopover* para aves limícolas na primavera é bastante reduzida quando comparada com o estuário do Tejo. As populações invernantes de Borrelho-de-coleira-interrompida, Borrelho-grande-de-coleira, o Pilrito-comum e a Rola-do-mar apresentaram uma tendência de decréscimo significativa na Caldeira de Tróia no período 2004-2012. Estes padrões não foram, no entanto, registados ao nível da totalidade do estuário para o mesmo período.

Wetlands, and particularly estuaries, are among the most threatened habitats for birds. Worldwide, human activities, both economical and recreational, are closely linked to habitat loss and degradation, accelerating waterbird population declines (Davidson & Rothwell 1993, International Wader Study Group 2003).

Despite the long-term monitoring of some wetlands (e.g. Rufino 1982, Costa & Rufino 1994, 1997), the overall status of Portuguese wader and other waterbird populations remains poorly known. However, a recent study has brought empirical evidence for worrying declines in the wintering populations of some of the most abundant wader species at the Tagus estuary and Ria de Aveiro (Catry et al. 2011). These results highlight the need to maintain long-term waterbird surveys, which can provide the first alerts for hampering biodiversity loss of wetlands. On the other hand, the importance of the Sado estuary (and several other Portuguese estuaries) as a stopover area for migratory shorebirds, is yet to be assessed. The Sado estuary is the second largest estuary in Portugal, and it has been classified as an internationally important wetland under the Ramsar Convention for supporting more than 1% of the wintering populations of Dunlin *Calidris alpina*, Ringed Plover *Charadrius hiaticula*, Avocet *Recurvirostra avosetta* and Black-tailed Godwit *Limosa limosa*, and of the breeding population of Black-winged Stilt *Himantopus himantopus*.

In the present study, we conducted monthly surveys of a small area of the Sado estuary, the Caldeira de Tróia, to assess the seasonal abundance of waterbirds and describe patterns of human disturbance. Nine-year trends of wintering waders at the Caldeira de Tróia are compared with counts carried out in the whole Sado estuary.

METHODS

» Study area

The Sado estuary, SW Portugal, comprises an area of ca. 180 km^2 of which 40% are tidal flats. This area is classified as Natural Reserve, Special Protection Area (SPA) and Important Bird Area (IBA). Caldeira de Tróia (38° 29.031'N, 8° 53.216'W) is a small tidal lagoon located at the end of the Tróia peninsula, connected to the estuary by an entrance at the north. Intertidal flats at the Caldeira cover ca. 1 km^2.

» Waterbird surveys and human disturbance

Seasonal abundances of waterbirds at the Caldeira de Tróia were determined through monthly bird counts carried during low tide (tide height<1.0), using a telescope from vantage points, between April 2004 and October 2012. Boat surveys were conducted annually during high-tide in January between 2004 and 2012 to count roosting birds in the whole Sado estuary area (including the Caldeira), as part of a monitoring program lead by Instituto da Conservação da Natureza e das Florestas (ICNF). Wintering population trends of the most common waders and gulls at the Caldeira de Tróia were assessed by averaging bird abundance during the winter period (November-February). For graphical purposes, the winter 2004/2005 is referred as 2005, 2005/2006 as 2006, and so on. These trends were then compared with those from the whole Sado estuary (January counts). Mediterranean gulls *Larus melanocephalus* started to be counted only in 2006 and trends are presented for the entire annual cycle, given that the wintering population at the Caldeira de Tróia is small (pers. data).

At the Caldeira de Tróia the number of shellfishers and bait-diggers was quantified in each visit (from 2005 onwards) as a proxy for human disturbance.

Spearman correlations were used to assess significant trends in waterbird populations and in the number of shellfishers and bait-diggers. All results are presented as mean ± standard deviation (SD). Computations were carried out using the R software (R Development Core Team, 2010).

RESULTS

» Composition and seasonal abundance of the waterbird community

The monthly average number of waterbirds recorded at the Caldeira de Tróia during the study period was 754 ± 661 (n =103 visits), belonging to 31 different species. Numbers of birds increase in the autumn/winter months (peaking in December; 1904 ± 435, n = 8) and are very low during late-spring and early summer (minimum numbers in June; 25 ± 14, n = 9).

Waders and gulls represent the most abundant groups of waterbirds during the entire annual-cycle. Dunlins are the most abundant wader during the winter but they are scarce or totally absent between March and September (Fig. 1). Sanderling *Calidris alba*, Ringed plover, Kentish plover *Charadrius alexandrinus*, Grey plover *Pluvialis squatarola*, Turnstone *Arenaria interpres*, Redshank *Tringa totanus* and Greenshank *Tringa nebularia* are regular occurrences at the Caldeira de Tróia almost all year round (Fig. 1), although peak abundances differ among species. While most species show smooth increasing numbers from the end of summer towards the winter, numbers of both Ringed and, particularly Kentish plovers, steeply increase in August (Fig. 1). Most species also show smooth decreases in their abundances following the end of winter, suggesting weak passage during spring migration (mostly in April-May). The exceptions to this phenological pattern is the Whimbrel *Numenius phaeopus* which occur in low numbers at the Caldeira but still show a peak during April-May (Fig. 1). Common sandpipers *Actitis hypoleucos* are regular at the Caldeira de Tróia (percentage of occurrence = 38.8%, n = 103 visits) but occur at very small numbers. Other wader species recorded occasionally (< 5% of all visits) during the study period include the Eurasian oystercatcher *Haematopus ostralegus*, Black-tailed godwit *Limosa limosa*, Eurasian curlew *Numenius arquata*, Little stint *Calidris minuta*, Red knot *Calidris canutus* and Curlew sandpiper *Calidris ferruginea*.

Among gulls, Lesser black-backed gull *Larus fuscus* is the most common species throughout the whole annual cycle, being particularly abundant during autumn and winter (Fig. 2). The Black-headed gull *Larus ridibundus* has a similar phenological pattern, while the Mediterranean gull is abundant during summer and autumn but mainly absent from the Caldeira during the remaining months (Fig. 2). Yellow-legged gulls *Larus michahellis*, although present in ca. 35% of all visits, occur at low numbers (maximum 10 individuals). Among terns, Sandwich tern *Sterna sandvicensis* are frequent (percentage of occurrence = 43.7%) mainly during autumn and winter, while Caspian tern *Sterna caspia*, Common tern *Sterna hirundo* and Little tern *Sterna albifrons* were recorded only once.

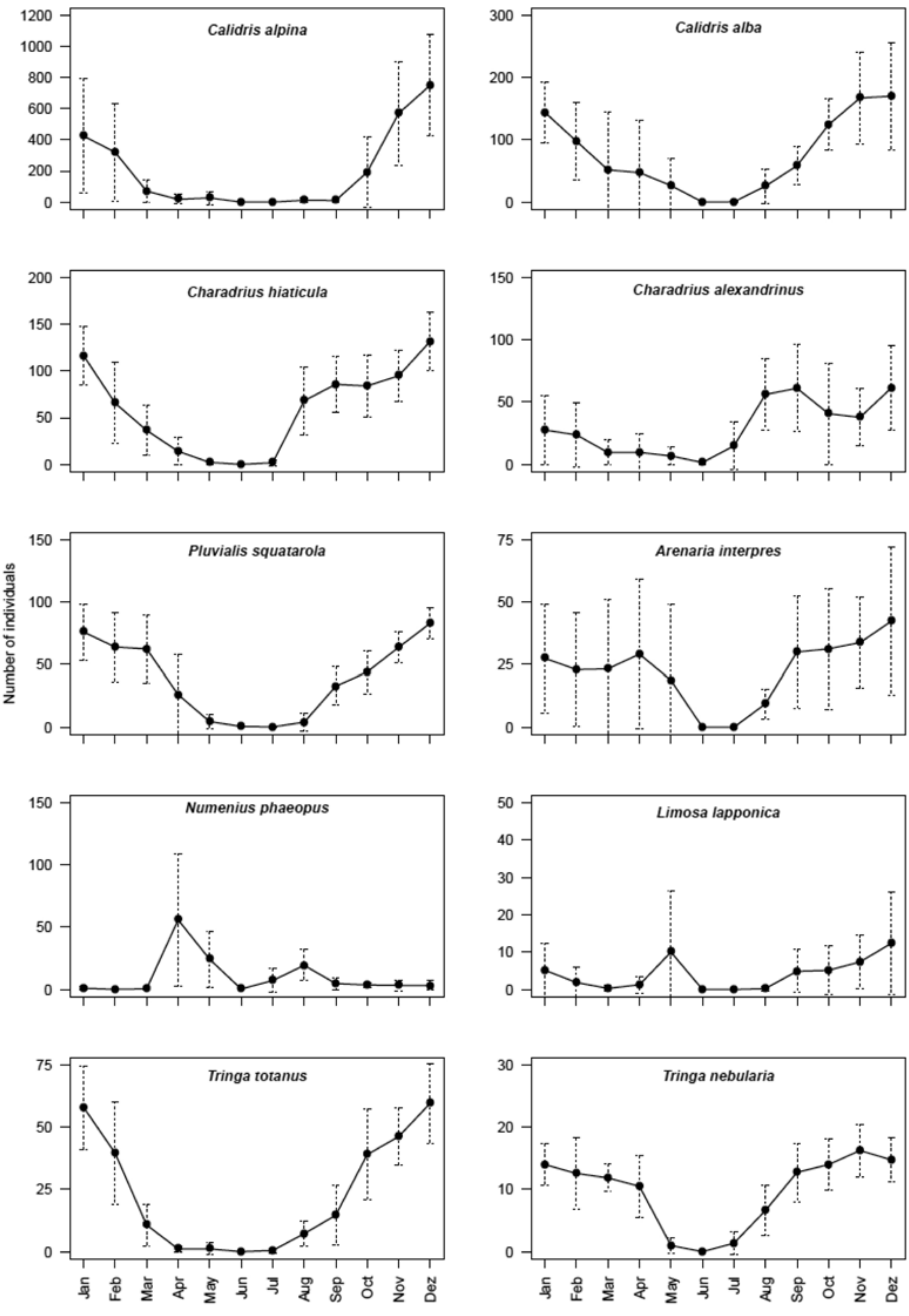

Figure 1. Phenology of the 10 most common wader species at the Caldeira de Tróia during the study period.
Figura 1. *Fenologia das 10 espécies de aves limícolas mais abundantes na Caldeira de Tróia no período de estudo.*

Grey heron *Ardea cinerea* and Little egret *Egretta garzetta* are regular species, present all year round at the Caldeira de Tróia (mean number of individuals per visit: 6.8 ± 4.2 and 4.2 ± 4.3, respectively). Great cormorants *Phalacrocorax carbo* are also regular visitors during the winter (mean number of individuals per visit in winter: 6.2 ± 7.1), as well as Red-breasted merganser *Mergus serrator* which occur in low numbers between November and March (mean number of individuals per visit: 4.0 ± 4.3). Greater flamingo *Phoenicopterus roseus* and Spoonbill *Platalea leucorodia* were recorded in ca. 12 and 3.9% of all visits, respectively.

» Trends of wintering waterbird populations

Four out of the 10 most abundant wintering wader species have significantly declined at the Caldeira de Tróia in the last decade, namely Dunlin (r = -0.738, p = 0.046), Ringed plover (r = -0.833, p = 0.015), Kentish plover (r = -0.833, p = 0.015) and Turnstone (r = -0.833, p = 0.015). In contrast, no declining trends were recorded at the whole Sado

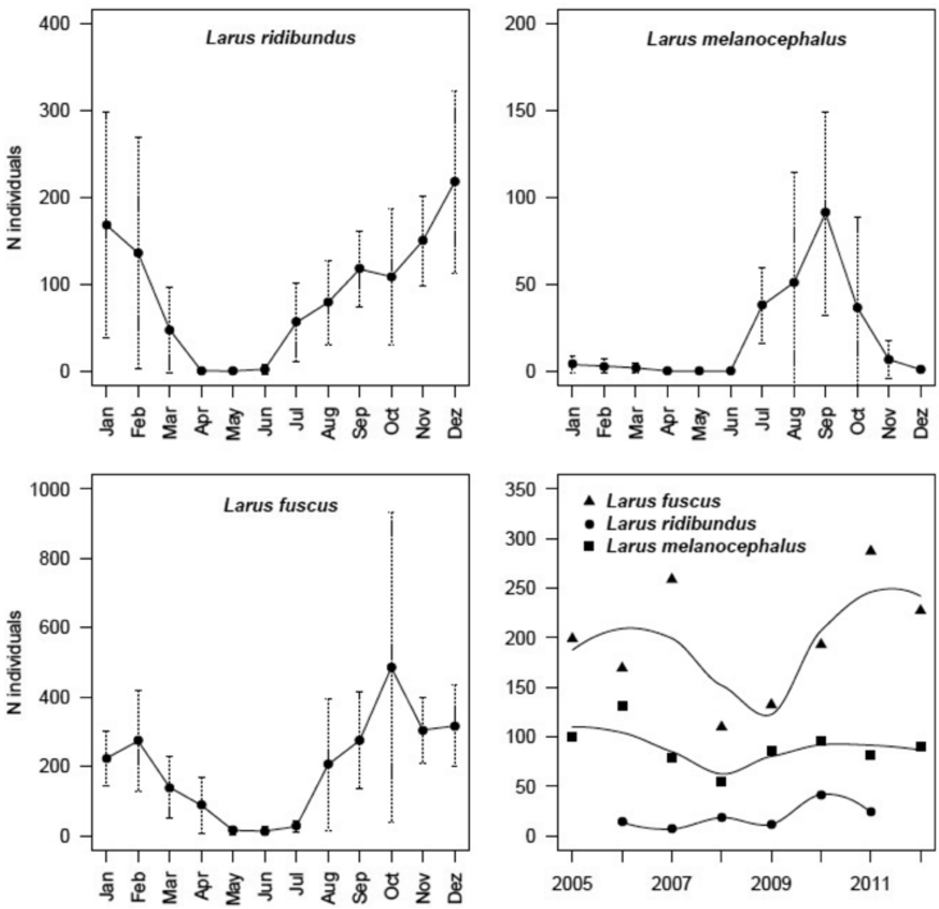

Figure 2. Phenology and population trends (last panel) of Black-headed gull (wintering trend), Mediterranean gull (annual trend) and Lesser black-backed gull (wintering trend) at the Caldeira de Tróia during the study period. Trends are represented by loess lines.

Figura 2. *Fenologia e tendências populacionais (último painel) de Guincho-comum (inverno), Gaivota-de-cabeça-preta (anual) e Gaivota-de-asa-escura (inverno) na Caldeira de Tróia durante o período de estudo. As tendências são representadas por curvas* loess.

estuary populations for these or other wader species (Fig. 3).

Among gulls, no significant trends were recorded at Caldeira de Tróia (Fig. 2).

» Human disturbance

An average number of 10.8 ± 11.0 (range = 0 to 73, n =103) shellfishers and bait-diggers per visit were recorded at the Caldeira de Tróia during the whole study period and 8.2 ± 5.5 during the winter. The peak of human activity was usually observed during the summer period, despite some annual variation. There were no significant trends in shellfishing and bait-digging activities during either the whole study period (r = -0.107, p = 0.840) and the winter period (r = -0.75, p = 0.07).

DISCUSSION

» Composition and seasonal abundance of the waterbird community

Caldeira de Tróia holds an important diversity of waterbirds throughout the year, although the winter is clearly the season with the highest diversity and bird abundance. The waterbird community of Caldeira de Tróia is very similar to that of the Sado estuary, except for the waterfowl, almost absent from the Caldeira (confirmed during regular high-tide visits; author's unpublished data) despite relevant international numbers recorded at the estuary-scale (Elias et al. 2006).

Although comprising less than 2% of the total estuary intertidal area, the Caldeira de Tróia holds proportionally large numbers of waders during the winter, particularly of Sanderling (28-204%, between 2004 and 2012), Greenshank (14-23%) and Turnstone (6-74%). The disproportional high numbers of these species at the Caldeira de Tróia could reflect a true preference for this area but might also be due to the underestimation of populations in high-tide counts, which might fail to accurately estimate low abundant species. The relative importance of the Caldeira tidal flats for other species was less systematic, but numbers of Dunlin, Ringed plover and Kentish plover reached >10% of total birds in some winters.

Abundance of waders at the Caldeira de Tróia during migratory periods, particularly in spring, is low, contrasting with the numbers recorded during the winter. Overall, wader numbers peak during the winter period and evidence for strong spring migration across the study area was provided only for Whimbrel, whereas Kentish plover uses the area mostly during late summer, either as a stopover or post-breeding dispersal area. The few existing data from surveys conducted outside the winter period at part (Alves et al. 2011) or at the whole (ICNF, unpublished data) estuary seems to support the low abundance of waders at the Sado estuary during spring migration. For instance, maximum numbers of Dunlins, Grey plovers and Ringed plovers counted in March and May 2011 (no available data for April) comprised ca. 7%, 44% and 0% of their mean wintering populations, respectively (n=405, 283 and 0; ICNF, unpublished data). Monthly surveys conducted at saltpans in the northern Sado estuary in 2010 also prove the absence of a peak abundance of waders during spring migration (Alves et al. 2011). This scenario is at variance to the one described by Catry et al. (2011) for the neighbour Tagus estuary, an important stopover area for several wader species, where peak abundances of birds are clearly noticeable during spring months. Tagus and Sado are the two largest estuaries in Portugal and those that historically harbour the larger wintering wader populations (e.g. e.g. Rufino 1982, Costa & Rufino 1994, 1997, Catry et al. 2011). Thus, the apparent contrasting value of the two estuaries for passage migrants in spring is particularly intriguing and certainly deserves further research. During migration, birds have higher energy demands and at stopover areas they face the need to balance conflicting demands between predator avoidance and food acquisition (e.g. Lindstrom 1990, Ydenberg et al. 2002). Apart from resource quality and availability, local habitat characteristics or landscape have proven to be critical factors in determining wader stopover choices (e.g. Sprague et al. 2008, Ydenberg et al. 2007). Open landscapes, which offer little cover for predators reducing their hunting efficiency, are often preferred by waders (Cresswell 1996, Sprague et al. 2008). The intertidal flats of the Tagus estuary occupy an area ca. 40% larger than those of the Sado estuary, which could explain the preference of migrating waders. In particular, the wide and open landscape of the northern mudflats of the Tagus estuary, providing

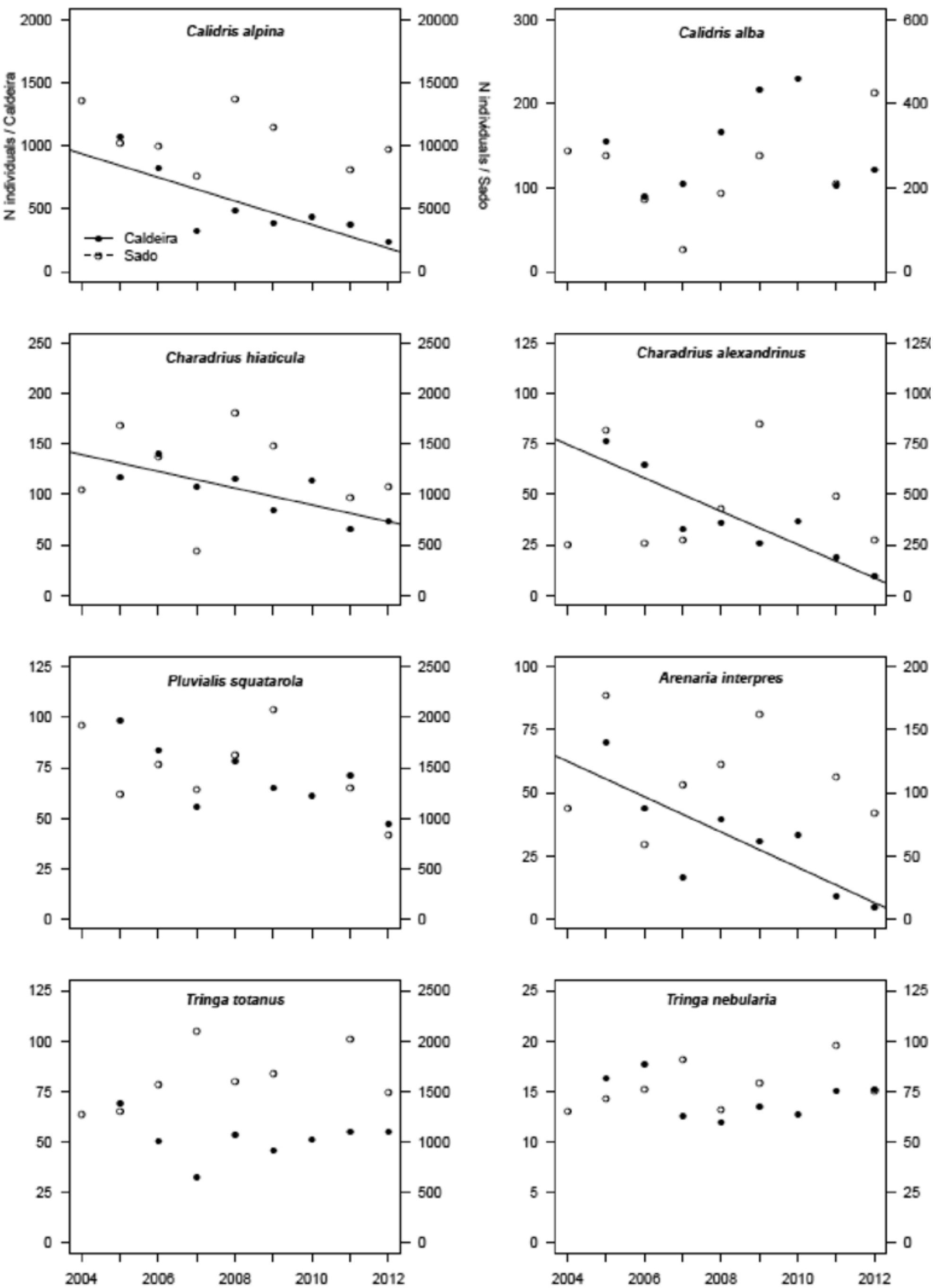

Figure 3. Wintering (November-February) population trends of the 10 most common wader species at the Caldeira de Tróia (black dots) and January trends of the same species at the whole Sado estuary (white dots) for comparative purposes. Significant trends ($p < 0.05$) for the Caldeira de Tróia are represented by a black line.

Figura 3. *Tendências das populações invernantes (Novembro-Fevereiro) das 10 espécies de aves limícolas mais abundantes na Caldeira de Tróia (pontos pretos) em comparação com as tendências registadas para as mesmas espécies em Janeiro em todo o estuário do Sado (pontos brancos). As tendências significativas ($p < 0.05$) para a Caldeira de Tróia são representadas por uma linha preta.*

foraging areas with large distances from cover, have no parallel at the Sado estuary. Observations from the Tagus estuary, where waders concentrate at the northern area during migration while being homogeneously distributed in winter (Catry et al. 2011) support this idea. During migration, waders invariably find themselves in unfamiliar surroundings and typically become more gregarious (Colwell 2010), which might also explain why some potentially high-quality areas attract low number of individuals.

Gulls were particularly abundant at the Caldeira de Tróia during autumn and winter, which replicates the pattern known for the Sado estuary (Elias et al. 2006). Mediterranean gulls are an exception to this pattern, being clearly more abundant during late summer and autumn at Caldeira de Tróia, although the Sado estuary has been listed among the most important wintering sites for the species in Portugal (Catry et al. 2010).

The regular occurrence of the Breasted merganser *Mergus serrator* and occasional occurrence of the Black-necked Greebe *Podiceps nigricollis* at the Caldeira de Tróia should be highlighted given that both species are relatively rare winter visitors in Portugal (Catry et al. 2010). The Sado estuary represents the most important wintering area for both species in Portugal (Elias et al. 2006).

» Trends of wintering waterbird populations

Declining numbers in the wintering populations of some wader species (Dunlin, Ringed plover, Kentish plover and Turnstone) at the Caldeira de Tróia were not consistent with the trends recorded at the Sado estuary scale for the same period. Changes in the quality of the foraging area of the Caldeira could potentially be involved in explaining such declines. While food availability was not assessed in this study, human disturbance did not show an increment during the study period (furthermore, human disturbance is greater during summer months when bird counts at the Caldeira are lowest). The decline in the population of Kentish plovers, can only be partly explained by the disappearance of a breeding population in the northern sandy beaches of Tróia peninsula (Catry & Catry 2011). The contrasting trends of these species at small and large scales clearly deserve more research. The large inter-annual variations in the number of wintering individuals of most species at the estuary scale suggest that longer periods might be necessary to capture real trends. On the other hand, changes in the availability of high quality areas at the estuary scale rather than the loss of quality of the Caldeira de Tróia could promote the redistribution of birds within the estuary, explaining the decreasing use of the Caldeira tidal flats.

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Population trends in the steppe birds of Castro Verde in the period 2006-2011: consequences of a drought event and land use changes?

Tendências populacionais das aves estepárias de Castro Verde no período 2006-2011: consequência de um evento de seca e de alterações no uso do solo?

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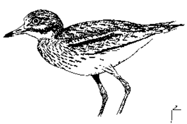

ABSTRACT - Castro Verde is the main area of cereal steppes in Portugal (ca. 80000 ha), having international importance for several steppe bird species with unfavourable conservation status. In spring 2006, a large-scale census of bird populations in the region was carried out. In the current paper, we (a) update the 2006 population estimates using correction for detectability, (b) report the results of a second census carried out in 2011, (c) explore the likely impacts of the 2005 drought event and land use changes on the observed population trends in the period 2006-2011. Correcting for the detectability bias of the 2006 counts resulted in an increase in population estimates ranging from 6% in the Stone Curlew to 210% in the Zitting Cisticola (mean= 75.4%, median = 61.4%, n=14 species), giving further significance to the importance of Castro Verde for the conservation of steppe birds in Portugal and Europe. Overall, between 2006 and 2011, species frequencies increased an average of 60% (median= 14%, range=-52.9% to 440.7%, n=14 species) and bird abundance increased 66% (median= 22.5%, range=-76.5% to 559.6%, n=14). This strongly suggests that bird populations in 2006 were still suffering the impact of the 2005 drought, and that the situation in 2011 is more representative of an average climatic context. The species registering the strongest population increases were the ones associated with cereal fields, in spite of the decreased availability of this habitat. This suggests that the 2005 drought was the main driver of a significant population crash and of the subsequent increasing population trend in spite of the ongoing habitat loss for this set of species.

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In contrast, the species registering the largest losses were the ones associated with sparse vegetation and bare ground, and for which the 2005 drought might have caused an increase in habitat quality.

RESUMO - A região de Castro Verde é a principal área de estepe cerealífera em Portugal (ca. 80000 ha) e possui importância internacional para várias espécies de aves estepárias com estatuto de conservação desfavorável. Durante a Primavera de 2006 foi efectuado um censo das populações de aves em toda a região. No presente artigo, (a) fazemos uma actualização das estimativas populacionais de 2006, aplicando um método que corrige a detectabilidade das espécies, (b) relatamos os resultados de um segundo censo efectuado em 2011, (c) exploramos os impactos potenciais da seca de 2005 e de alterações do uso do solo nas tendências populacionais observadas no período 2006-2001. A correcção das estimativas de 2006 resultaram num aumento das estimativas populacionais que variou entre 6% no Alcaravão e 210% na Fuínha dos juncos (média= 75,4%, mediana = 61,4%, n=14 espécies), dando ainda mais ênfase à importância de Castro Verde para a conservação das aves estepárias em Portugal e na Europa. Globalmente, entre 2006 e 2011, as frequências de ocorrência das espécies aumentaram em media 60% (mediana= 14%, limites de variação=-52,9% a 440,7%, n=14 espécies) e as abundâncias aumentaram 66% (mediana= 22.5%, limites de variação =-76,5% a 559,6%, n=14). Este resultado sugere fortemente que as populações de aves em 2006 ainda estavam a sofrer os efeitos da seca de 2005, e que a situação em 2011 é mais representativa de um contexto climático normal. As espécies que registaram os maiores aumentos populacionais foram as associadas aos campos de cereal, apesar do decréscimo da área deste habitat durante o período de estudo. Isto sugere que a seca de 2005 foi a principal causa do declínio populacional e da subsequente recuperação, mesmo num contexto de perda de habitat. Em contraste, as espécies que registaram as maiores perdas populacionais foram as associadas a vegetação esparsa e solo nú, e para as quais a seca de 2005 poderá ter representado um aumento da qualidade do habitat disponível.

Castro Verde is the main cereal steppe area in Portugal and has national and international importance for populations of several steppe bird species including Great Bustard *Otis tarda*, Little Bustard *Tetrax tetrax*, Calandra Lark *Melanocorypha calandra*, Lesser Kestrel *Falco naumanni*, Stone Curlew *Burhinus oedicnemus*, Roller *Coracias garrulus* and Black-Bellied Sandgrouse *Pterocles orientalis* (Costa et al. 2003). In 2006, as part of a EUFAR (European Fleet for Airborne Research) research project (STEPPEBIRD), the first large-scale assessment of distribution patterns and breeding population estimates for 16 steppe bird species in the region was carried out (Moreira et al. 2007). This census produced a baseline characterisation against which the results of future bird monitoring, using similar methodology, could be compared.

The previous estimates of bird abundance for the Castro Verde region in 2006 were based on raw count data, i.e. unadjusted for the imperfect detection of birds. As such, they were acknowledged as minimum population estimates (Moreira et al. 2007). The detection of all birds at a sample location is impossible to achieve due to the increasing difficulty of detection as distance from the observer increases (Buckland et al. 2001). This means that abundance estimates derived directly from count data are only relative rather than absolute measures (Buckland et al. 2008), potentially resulting in underestimation of population size. Use of abundance estimates adjusted for detectability therefore gives more robust estimates of population trends than use of raw density estimates (Norvell et al. 2003).

A second census was carried out in 2011, and its results constitute the bulk of the current paper. Of particular interest is the fact that the 2006 census took place in the spring just after the worst drought for 60 years in Portugal. In fact, during 2005 rainfall in the region was just 40% of the annual average (INAG 2005) which resulted in a poor agricultural year, particularly for dry crops (cereal yield was very low, with many failed crops). Monthly rainfall patterns from November 2004 to September 2005 were well below the levels of an average year. Consequently, in 11 of the 12 months of the hydrological year 2004/2005 the situation was classified as "severe or extreme drought". This drought probably had important impacts on bird populations, mainly for resident species, which were likely reflected in the population estimates. The year 2011, in contrast, was considered an average year in terms of rainfall and temperature (Instituto de Meteorologia 2011). Thus, under a global change scenario where Portugal is likely to be affected by more frequent and intense heat waves (Santos et al. 2001), this comparison of the two censuses may yield useful insights into the impact of future droughts on steppe birds. During the same time period, farming management changes as a result of agricultural policies may also have caused impacts on bird populations. Therefore, the objectives of the current paper are threefold: (a) to update the previous population estimates using more refined techniques; (b) to report the results of a second census carried out following the recommendations of the initial study, and; (c) to evaluate the potential impacts of a drought event and land use changes on the observed population trends in the period 2006-2011.

METHODS

» Study area

The Castro Verde Special Protection Area (SPA) is a plain (altitude 100–300 m) of about 80000 ha, with a Mediterranean climate characterised by hot summers (30–35ºC on average in July), fairly cold winters (averaging 5-8ºC in January) and over 75% of annual rainfall (500–600 mm) concentrated in October–March (Delgado & Moreira 2000, Moreira et al. 2005). It is mainly occupied with pseudo-steppe habitats (fallow, pastures, cereal fields, ploughed land) created by farming activities. In the north and south of the region there are Holm Oak *Quercus rotundifolia* woodlands (montados) offering sparse tree cover, frequently with a grassy understorey grazed by livestock. Areas of shrubland occur mainly in association with river valleys and in the southeastern part of the region. For further details, see Moreira et al. (2007).

» Sampling scheme

The sampling area corresponded to the core of the SPA, a rectangle of 44860 ha where pseudosteppe habitat prevailed (Fig. 1). Our sampling scheme consisted in a grid of 370 sampling points placed throughout the study area in a systematic manner, by assigning one sampling point to each GAUSS 1×1km grid square (Hayford-Gauss projection, International Ellipsoid, Datum Lisboa IGeoE) (Fig. 1). The sampling points were located over dirt tracks (for accessibility) and as close as possible to the square's centre. A 125 m circular buffer was defined around each point, and it was also required that this buffer fell completely on pseudo-steppe habitat and within a single grid square. In cases where these conditions did not apply, the grid square was not surveyed. This same methodology was applied in the 2006 census, but because of changes in land cover and accessibility, the resulting set of points was not the same although both were valid samples of the habitats available.

» Bird counts

Bird censuses were carried out at the selected sampling points using 5-minute point counts with a distance limit of 125 m (Fuller & Langslow 1984, Bibby et al. 2000). All registrations within the buffer were recorded, along with the estimated distance to the centre of the buffer (to the nearest 5 to 10 m). Whenever possible, the sex and age group (juvenile or adult) of the birds was recorded. Bird counts were carried out between the 15 April and 15 May 2011 (with the exception of two points censused during the first week of April) by 22 observers. All counts were carried out in the first 4 hours after sunrise and in the last 2 hours before sunset. Categorisation to the genus level was made for the Crested and Thekla larks (*Galerida cristata* and *Galerida theklae*) due to difficulties in reliably identifying all individuals of these two species in the field. All observers were experienced, and a practical session to improve accuracy in estimating distances was carried out prior to the surveys.

» Habitat measurements

Land-use information was collected in each sampling point by dividing the 125m-radius buffer into 8 quadrants and visually estimating the dominant habitat in each one of them. The following habitat categories, corresponding to the major land uses in the rotational dry crop system in the region, were considered: a) fallow land and grasslands; b) cereal fields; d) ploughed fields. For each of these habitats, an index of availability ranging from 0 (habitat not dominant in any quadrant) to 8 (habitat dominant in all quadrants) was calculated. The occurrence of scrub patches within the buffer was also registered.

» Data analysis

For the purposes of this study, we selected 14 species (Table 1) including mostly ground-nesters, but also non-obligate ground-nesters that, although not exclusive to steppe-like habitats, were fairly common in the study area. The Great Bustard and Black-Bellied Sandgrouse were not included, as the method used is not the most appropriate to census these species (Moreira et al. 2007). For species with a minimum sample size of 40 observations pooled across both censuses, the program DISTANCE (Thomas et al. 2010) was used to estimate detection functions and to correct population estimates for detectability. Only two species did not attain this minimum required sample, the Black-eared Wheatear *Oenanthe hispanica* and the Tawny pipit *Anthus campestris*. To minimise the detectability

Table 1. List of the 14 studied species, frequency of occurrence in 2006 (n= 391 points) and 2011 (n=370 points), proportional variation 2006-2011, and results of the binomial test to compare proportions (ns = non-significant, *= p<0.05, **=p<0.01, ***p<0.001).

Tabela 1. *Lista das 14 espécies estudadas, frequência de ocorrência em 2006 (n= 391 pontos) e 2011 (n=370 pontos), variação proporcional no período 2006-2011, e resultados do teste binomial para comparar proporções (ns = não significativo, *= p<0.05, **=p<0.01, ***p<0.001).*

Scientific name	Common name	Frequency 2006	Frequency 2011	Δ(%) 2006-2011	Binomial test
Emberiza calandra	Corn Bunting	0.783	0.849	+8.4	*
Melanocorypha calandra	Calandra Lark	0.294	0.392	+33.3	**
Galerida spp.	*Galerida* larks	0.289	0.408	+41.2	**
Tetrax tetrax	Little Bustard	0.276	0.330	+19.6	ns
Saxicola torquata	Stonechat	0.148	0.151	+2.0	ns
Cisticola juncidis	Zitting Cisticola	0.113	0.611	+440.7	***
Alectoris rufa	Red-legged Partridge	0.100	0.170	+70.0	**
Calandrella brachydactyla	Short-toed Lark	0.097	0.068	-29.9	ns
Circus pygargus	Montagu's Harrier	0.066	0.103	+56.1	ns
Coturnix coturnix	Quail	0.066	0.273	+313.6	***
Burhinus oedicnemus	Stone Curlew	0.064	0.054	-15.6	ns
Upupa epops	Hoopoe	0.061	0.062	+1.6	ns
Anthus campestris	Tawny Pipit	0.054	0.035	-35.2	ns
Oenanthe hispanica	Black-eared Wheatear	0.051	0.024	-52.9	ns

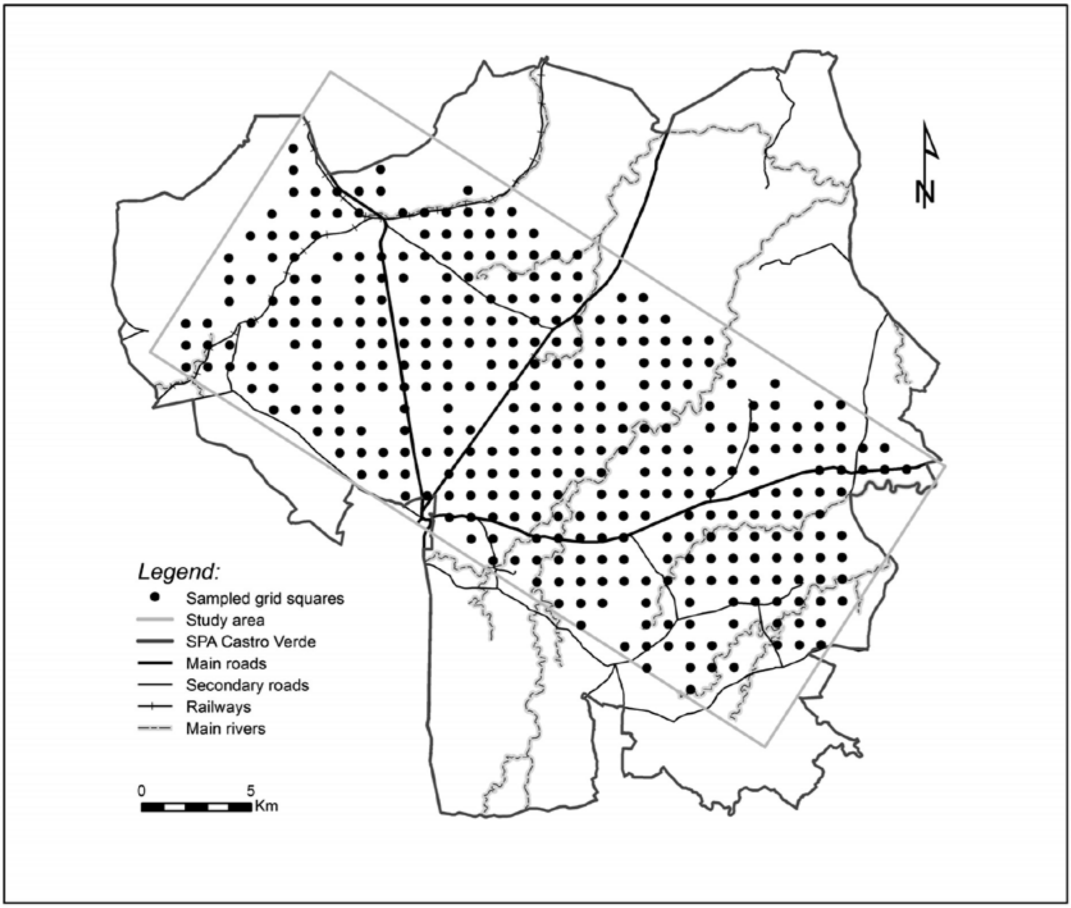

Figure 1. Study area (rectangle) and location of survey points for bird counts.
Figura 1. *Área de estudo (rectângulo) e localização dos pontos de amostragem das aves.*

bias, the data from these species were pooled and a common detectability function was estimated. Use of the Distance sampling methodology is based on some assumptions, in particular: (1) birds at the centre of the point are always detected; (2) birds are detected at their initial location, prior to any movement in response to the observer; (3) distances are measured accurately, or are correctly allocated to distance intervals (Thomas et al. 2010).The count period used at each point is an important factor for meeting assumption (2). If counts are too long then birds may be double counted, but if counts are too short then birds may be missed. Fuller & Langslow (1984) found that surveys of between 5 and 10 minutes in length were best for maximising bird sightings whilst also minimising the double counting of birds and the potential for movement of birds into the study area during the count. For all three assumptions, it is important that skilled observers are used; all observers were experienced in bird identification and trained in how to conduct distance sampling surveys.

While distance data were collected to the nearest 5 m with a maximum distance of 125 m, for analysis these values were transformed into intervals with cut points at 0, 20, 40, 60, 80, 100, 120 m. It was important to place the distance data into intervals because of the variability in distance estimates by different observers. This difference in accuracy created data spikes that reflect differences between observers rather than true variation in detectability. Use of 20 m intervals was found to provide a smoother graph of the frequency of distance measurements. DISTANCE uses the midpoint of these intervals for analysis. In accordance with the guidance of Buckland et al. (2001) that the 10% of detections associated with the largest distances should be discarded, distances were truncated at 120 m, meaning that any observations at 120 m or farther were removed from the analysis. The selection of truncation distance is a balance between the fact that larger truncation distances will result in small tail probabilities that increase bias, and the fact that smaller truncation distances mean more data is discarded, reducing precision. Distance sampling methodology has been shown to be robust to right truncation (Ekblom 2010). Truncation of distances also has the advantage that it reduces the complexity of the detection function, generally allowing for a better fit to the remaining data (Dallimer et al. 2009). Conventional distance sampling models were run for each species individually (except for Crested/Thekla Larks, and Tawny Pipit/Black-eared Wheatear, as mentioned above). Half-normal and hazard-rate functions were used to fit the detection function, with the best model chosen by Akaike's Information Criterion (Burnham et al. 2002). Adjustment terms were not used since they can lead to over-fitting, especially when sample size is small. Observations from both years were used to generate the detection function, but estimations of abundance were stratified by year.

The detection functions for Quail *Coturnix coturnix* and Little bustard *Tetrax tetrax* were both best modelled by hazard-rate functions. For all other species in the study, half-normal functions gave the best models. The hazard-rate function allows for a better fit when the data has a larger shoulder (i.e., for species that do not have a sharply decreasing detectability as distance from observer increases).

The DISTANCE-corrected estimates were based on: (a) the number of males for songbirds (Passeriformes), and assuming one male is equivalent to a breeding couple, (b) the number of males for the Little Bustard; (c) the total number of individuals for the remaining species where separation between males and females is difficult, or when the total population is usually assessed without discriminating sex. The total steppe area in the Castro Verde region is 55490 ha (Moreira et al. 2007), so bird abundance estimates were extrapolated to this entire area.

Differences in the frequency of occurrence (defined as the number of count points where a species was recorded, expressed as a proportion of the total number of points) across years were tested for each species using the binomial test to compare proportions (Crawley, 2013). The same test was used to compare shrub patch prevalence. Differences in the index of habitat availability for the three habitat types was compared across years using the non-parametric Wilcoxon test (Crawley, 2013). Finally, bird population estimates across years were compared using a z-test using the DISTANCE estimates of means and standard deviations.

RESULTS

The frequency of occurrence of each species and its population estimates are shown in Tables 1 and 2, respectively. Overall, between 2006 and 2011, species frequencies increased an average of 60% (median= 14%, range=-52.9% to 440.7%, n=14 species) and bird abundance increased 66% (median= 22.5%, range=-76.5% to 559.6%, n=14). The more significant increases in frequency and abundance were registered for the Zitting Cisticola *Cisticola juncidis*, Quail *Coturnix coturnix* and Red-legged Partridge *Alectoris rufa*, with over 70% increases in frequency and over 60% increases in abundance. In contrast, major declines (over ca.30% in prevalence and abundance) were registered for the Short-toed Lark *Calandrella brachydactyla*, Tawny Pipit *Anthus campestris* and Black-eared Wheatear *Oenanthe hispanica*. The remaining species showed moderate and variable (positive or negative) trends. Differences in the availability of different habitats between 2006 and 2011 were particularly evident in the case of cereal fields, which decreased ca. 34% (Table 3). A ca. 10% near-significant increase in the availability of fallow land was also registered. The proportion of points with shrubland patches did not differ significantly between years (11.7% and 13.0%, respectively for 2006 and 2011; chi-square test=0.16, p=0.692).

Table 2. Population estimates for Castro Verde steppe birds. For each species, the 2006 estimates uncorrected for detectability (Moreira et al. 2007), the 2006 DISTANCE corrected estimates and 2011 DISTANCE corrected estimates are given. Units for the estimates are given as breeding pairs, males or total number of birds. Values are shown as average (and 95% confidence intervals). The significance of the z-test to compare years is also shown (ns= não significativo, *=p<0.1, ***=p<0.01).

Tabela 2. *Estimativas populacionais das aves estepárias em Castro Verde. Para cada espécie são indicadas as estimativas de 2006 não corrigidas (Moreira et al. 2007) e com a correcção para a detectabilidade (usando o DISTANCE) e as estimativas corrigidas para 2011. As unidades são, dependendo da espécie, casais, achos ou o número total de indivíduos. Os valores indicados são médias (e intervalos de confiança a 95%)(ns= não significativo, *=p<0.1, ***=p<0.01).*

Scientific name	unit	2006 uncorrected	2006 corrected	2011 corrected	Δ(%) 2006-2011	Z-test
Emberiza calandra	pairs	16185 (14582-17519)	17940 (15652-20561)	23540 (20674-26803)	+31.2	***
Melanocorypha calandra	pairs	6160 (4910-7410)	12801 (9699-16895)	16250 (12810-20616)	+26.9	***
Galerida spp.	pairs	3734 (3031-4437)	8682 (6450-11686)	13854 (10803-17767)	+59.6	***
Tetrax tetrax	males	4213 (3402-5025)	4774 (3743-6089)	5211 (4126-6583)	+9.2	***
Saxicola torquatus	pairs	1540 (1122-1958)	2740 (1691-4440)	3238 (2032-5161)	+18.2	***
Cisticola juncidis	pairs	1308 (931-1684)	4052 (2897-5669)	26726 (21987-32486)	+559.6	***
Alectoris rufa	birds	1511 (1022-2000)	2190 (1396-3437)	3546 (2375-5295)	+61.9	***
Calandrella brachydactyla	pairs	1453 (939-1966)	4116 (2449-6918)	2940 (1750-4938)	-28.6	***
Circus pygargus	birds	872 (521-1223)	1074 (613-1883)	1670 (961-2902)	+55.5	***
Coturnix coturnix	birds	930 (553-1307)	1144 (735-1779)	4193 (3254-5404)	+266.5	***
Burhinus oedicnemus	birds	988 (579-1397)	1046 (533-2052)	872 (437-1737)	-16.6	ns
Upupa epops	birds	872 (485-1258)	1812 (1021-3218)	1737 (972-3104)	-4.1	ns
Anthus campestris	pairs	465 (260-670)	947 (486-1844)	223 (81-614)	-76.5	***
Oenanthe hispanica	pairs	552 (302-802)	613 (296-1268)	390 (169-899)	-36.4	*

Table 3. Habitat availability index in 125m- buffers around points used for bird censuses in 2006 and 2011. This index ranges from a minimum of zero to a maximum of 8 (corresponding to 100% cover; see methods). Values are shown as average ± standard deviation (upper line) and median (and 25% and 75% quartiles). The result of the Wilcoxon test to compare years is also shown (ns= non-significant, *=p<0.05, **=p<0.01).
Tabela 3. *Índice de disponibilidade de habitats nos pontos de contagem, em 2006 e 2011. Este índice varia entre um mínimo de zero e um máximo de oito (correspondendo a 100% de cobertura; ver metodologia). Os valores indicados são médias ± desvio padrão (linha superior) e medianas (e quartis 25% e 75%) (linha inferior). É tambem apresentado resultado do teste não paramétrico de Wilcoxon na comparação entre anos (ns= não significativo, *=p<0.05, **=p<0.01).*

Habitat	Availability 2006	Availability 2011	Wilcoxon Test's *P*
Fallow land	4.52±3.29 5(0-8)	4.97±3.16 6(2-8)	*
Cereal fields	2.33±2.94 0(0-4)	1.74±2.56 0 (0-4)	**
Ploughed fields	0.10±0.73 0(0-0)	0.12±0.67 0 (0-0)	ns

DISCUSSION

A few assumptions of the DISTANCE estimation method could not be strictly followed; most noticeably points were not randomly located. However, a systematic survey design, as recommended (Thomas et al. 2010), was used, and the resulting set of sampled points is a representative sample of the habitats available in the area. Additionally, the assumption of independence of detection is not straightforward for all the studied species, e.g. the Montagu´s Harrier tends to be a colonial or semi-colonial species and the Little Bustard is thought to have a lek breeding system where males can be clustered. Fortunately, distance sampling is robust to failures of the independence of detection (Thomas et al. 2010).

The effects of the correction for detectability bias within the 2006 counts resulted in an increase in population estimates ranging from 6% in the Stone Curlew to 210% in the Zitting Cisticola (mean= 75.4%, median = 61.4%, n=14 species), giving further significance to the importance of Castro Verde for the conservation of steppe birds in Portugal and in Europe.

Overall, in comparison with 2006, in 2011 we recorded a substantial increase (more than 60%) in species frequencies and abundances. This strongly suggests that bird populations in 2006 were still suffering the impact of the 2005 drought (e.g. through mortality or dispersion to more suitable areas), the worst drought of the last 60 years in Portugal, and that the situation in 2011 is more representative of average climatic conditions. A substantial (more than 50%) population increase was registered for species associated with taller and denser vegetation, namely the ones specializing in cereal fields, such as the Zitting Cisticola, Quail and Montagu´s Harrier (e.g. Borralho et al. 1998, Delgado & Moreira 2000, Millon et al. 2002, Moreira et al. 2007). The Corn Bunting, also associated with cereal fields, increased its population by more than 30%. In the 2005 drought year, the vegetation of most cereal crops did not develop, to the extent that farmers introduced livestock grazing in failed cereal crops (Moreira et al. 2007). It could thus be expected that species more dependent on this habitat type were among the ones suffering the most negative impact as a consequence of the drought. Other bird censuses at the national level have suggested a strong impact of the drought of 2004/2005 on bird populations (Hilton 2006). In contrast to the gains shown by most bird species, a few registered strong declines over the 2006-2011 period, most noticeably Tawny Pipit (-76%), Black-Eared Wheatear (-36%) and Short-Toed Lark (-29%). As this set of species is associated with sparse vegetation and bare ground

(Delgado & Moreira 2000, Moreira et al. 2007), we might expect overall habitat suitability for these to have increased in the drought year, and this might have explained the observed decrease by 2011. Given the fact that these species are trans-Saharan migrants, climatic and habitat conditions in their wintering grounds can also contribute to the observed population trends.

In addition to the drought effect, other land use changes currently taking place are expected to have impacted bird populations during the study period. In fact, significant changes to the Common Agricultural Policy (CAP) were introduced in the 2003 CAP reform, in particular the Single Farm Payment and the associated decoupling of payments from production (Oñate et al. 2007, Tranter et al. 2007). Decoupling means that farmers are no longer required to maintain production for receiving CAP payments, but only to keep land in good environmental and agricultural conditions (e.g. Brady et al. 2009). Moreira et al. (2007) hypothesised that, as a consequence of decoupling, dry cereal cultivation would be progressively abandoned, and this was expected to represent habitat loss for Corn Bunting, Zitting Cisticola, Quail and Montagu's Harrier. In fact, the estimated area occupied by cereal fields declined ca. 30% from 2006 to 2011, and therefore a decline could be expected in the populations of these species. The fact that the observed trend was exactly the opposite suggests that the 2005 drought was the main driver of a significant population crash and of the subsequent increasing population trend in spite of the ongoing habitat loss. Other species hypothesized to decline with the end of cereal cultivation were the ones associated with ploughed land, such as the Black-eared Wheatear, Short-toed Lark and Tawny pipit, because in the absence of cereal cultivation, tilling would no longer be needed to prepare fields for sowing (Moreira et al. 2007). However, the estimated area of ploughed fields did not vary significantly between 2006 and 2011. As a result of the decline of cereal cropping in the region, the area of permanent pastures has been increasing during the last decades (Delgado & Moreira 2010) and this trend continued in the period 2006-2011, with an estimated 10% increase. This might have contributed to explain the observed increases in species associated with fallows and pastures, such as the Little Bustard and the Calandra Lark (e.g. Moreira et al. 2007, Morgado et al. 2010). Finally, Moreira et al. (2007) hypothesised that agricultural abandonment and subsequent scrub encroachment were expected, in the medium term, to improve habitat suitability for species such as *Galerida* larks, and Red-legged Partridges. Both species were observed to have increased considerably, although shrubland prevalence in the sampled points did not differ significantly.

For species for which data from population trends exist from other sources, the current study corroborates the existing evidence in a few cases. For the Little Bustard, the long-term population monitoring described in Delgado & Moreira (2010) also shows a moderate increase in the studied period (*unpublished data*). Published data for the period 2004-2009 at the national level, coming from the Common Breeding Bird census in Portugal (Meirinho et al. 2011), suggests a strong increase in the population of Crested Lark consistent with the observed trends in Castro Verde. In contrast, inconsistent or opposite trends have been registered for Quail (moderate decline at the national scale), Zitting Cisticola (uncertain trend) and Corn Bunting (uncertain trend) (Meirinho et al. 2011), suggesting that the results observed in Castro Verde reflect mainly a local/regional trend.

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