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Action on multiple fronts, illegal poisoning and wind farm planning, is required to reverse the decline of the Egyptian vulture in southern Spain

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A B S T R A C T

Large body-sized avian scavengers, including the Egyptian vulture (*Neophron percnopterus*), are globally threatened due to human-related mortality so guidelines quantifying the efficacy of different management approaches are urgently needed. We used 14 years of territory and individual-based data on a small and geographically isolated Spanish population to estimate survival, recruitment and breeding success. We then forecasted their population viability under current vital rates and under management scenarios that mitigated the main sources of non-natural mortality at breeding grounds (fatalities from wind farms and illegal poisoning). Mean breeding success was 0.68 (SD = 0.17) under current conditions. Annual probabilities of survival were 0.72 (SE = 0.06) for fledglings and 2 yr old non-breeders, 0.73 (SE = 0.04) for non-breeders older than 2 yrs old and 0.93 (SE = 0.04) for breeders. Probabilities of recruitment were 0 for birds aged 1–4, 0.10 (SE = 0.06) for birds aged 5 and 0.19 (SE = 0.09) for older birds. Population viability analyses estimated an annual decline of 3–4% of the breeding population under current conditions. Our results indicate that only by combining different management actions in the breeding area, especially by removing the most important causes of human-related mortality (poisoning and collisions on wind farms), will the population grow and persist in the long term. Reinforcement with captive breeding may also have positive effects but only in combination with the reduction in causes of non-natural mortality. These results, although obtained for a focal species, may be applicable to other endangered populations of long-lived avian scavengers inhabiting southern Europe.

1. Introduction

Forecasting the fate of threatened populations and designing adequate conservation measures is one of the greatest challenges for scientists and conservation managers (Morris and Doak, 2002). Population viability analysis (PVA) is a key tool to model population dynamics, estimate extinction probabilities and evaluate the adequacy of different management strategies for maximizing population persistence (Caswell, 2001; Morris and Doak, 2002). To project realistic future population numbers and make credible conservation diagnoses of threatened species, PVA requires robust data on population size and/or demographic parameters (Morris and Doak, 2002). Age- and stage-dependent patterns of survival and reproduction determine the shape of the reproductive value function (McNamara and Houston, 1996), which ultimately drives population growth rates (Caswell, 2001; Morris and Doak, 2002).

The estimation of age- and stage-dependent demographic parameters in long-lived species requires individual long-term monitoring programs (Grande et al., 2009; Hernández-Matías et al., 2013; Sanz-Aguilar et al., 2009). This is the case for long-lived territorial raptors, in which individual marking and monitoring of both territorial and non-territorial birds require extraordinary field

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effort. Consequently, estimates of age-dependent vital rates within a species are typically available for only a limited number of populations and are used to apply PVA to different populations or closely related species (Carrete et al., 2009; Hernández-Matías et al., 2013; Martínez-Abraín et al., 2012). However, the results obtained from a single population cannot be taken as characteristic of the species or exported to other populations without caution (Sanz-Aguilar et al., 2009). Inter-population differences in demographic parameters, such as reproduction, survival or age-dependent recruitment, may exist even for very close populations (Sanz-Aguilar et al., 2009). Consequently, to robustly forecast population fates and evaluate conservation actions, an improvement in the current knowledge of inter-population variability in demographic parameters of endangered species is urgently needed (Hernández-Matías et al., 2013).

Due to their rapid decline worldwide and the ecosystem services they provide, scavenger vultures are considered priority species for conservation (Directive 2009/147/EC of the European Union on the Conservation of Wild Birds). As with other raptor species, vultures are threatened by multiple human factors including persecution (poisoning), habitat destruction, changes in sanitary policies and agricultural practices and fatalities due to infrastructures (Ogada et al., 2012). Reintroduction or reinforcement programs based on demographic data and PVA analyses have been carried out for several species (e.g., white-tailed eagles *Haliaeetus albicilla* Green et al., 1996; California Condors *Gymnogyps californianus* Meretsky et al., 2000; or Griffon vultures *Gyps fulvus* Sarrazin and Legendre, 2000). However, the introduction of individuals should be implemented only after environmental causes of decline or potential threats have been dealt with (Ewen et al., 2012; Pérez et al., 2012).

The Egyptian vulture (*Neophron percnopterus*) is a globally endangered long-lived scavenger that has experienced a severe population decline throughout its range (BirdLife International, 2007). The Spanish population which comprises ca. 80% of European population and has declined by 25% in the last two decades (Donázar, 2004) due to high non-natural mortality rates (Iñigo et al., 2008). Thus, conservation actions are urgently needed. The main causes of non-natural mortality (i.e., poisoning, electrocution, collision and direct-persecution) have been identified (Hernández and Margalida, 2009; Grande et al., 2009; Cortés-Avizanda et al., 2009) and in some cases roughly quantified (i.e., wind farm mortality, Carrete et al., 2009). Poisoning and wind farm mortality mainly affect territorial breeders (Carrete et al., 2009; Hernández and Margalida, 2009). This is especially relevant for a long-lived species in which population growth rates are expected to be highly sensitive to changes in adult survival (Sæther and Bakke, 2000). On the other hand, a European studbook for the Egyptian vulture and the European Endangered Species Program (EEP) for this species is being developed in order to reinforce and/or reintroduce populations from captive breeding programs (http://www.zoopraha.cz/en/animals/we-help-them-to-survive/projects/7687-egyptian-vulture-conservation-in-the-balkans; see also BirdLife International, 2007; http://www.capovaccaio.it).

This study focuses on the most southern European population of Egyptian vultures (Andalusia, Spain), where numbers of Egyptian vultures have dropped precipitously during the last few decades to the current <25 pairs isolated from other populations (Carrete et al., 2009). Although the dynamics of the species in terms of the number of breeding territories occupied are well known (Carrete et al., 2007; Donázar, 2004), detailed estimates of survival parameters are only available for the Ebro population in Northern Spain (Grande et al., 2009) and there are no age-dependent estimates of recruitment. These deficiencies lead to important uncertainty in population viability analyses (Tauler et al., 2015), which directly affect the design of management actions for the conservation of the species on a broad scale.

Our specific objectives were to: (1) assess the viability of the southern Spanish population of Egyptian vultures through the estimation of local demographic parameters (survival, recruitment and breeding success); (2) evaluate the potential existence of inter-population variability in demographic parameters for the species by comparing the estimated vital rates with available estimates (Grande et al., 2009); and (3) forecast population viability by considering different management approaches, particularly the mitigation of human-induced mortality associated with wind farms and poisoning and the release of different numbers of fledglings to reinforce the population.

2. Methods

2.1. Species and study area

The Egyptian vulture is a highly opportunistic species, foraging on small wild prey and on carcasses of small and medium-sized animals (Donázar, 1993). Non-breeders typically aggregate around predictable food resources (e.g., landfills or vulture restaurants, so-called "muladares") during the breeding season (Donázar et al., 1996; López-López et al., 2013). Individuals acquire adult plumage and are able to reproduce at five years of age (Carrete et al., 2009). They tend to recruit into territories near their natal areas and, once established, adult breeders are extremely philopatric to their breeding territories (Donázar, 1993; Carrete et al., 2007). Continental Western European populations of Egyptian vultures are migratory, with birds crossing the Sahara and spending the wintering season (and sometimes their first year of life) in the sub-Saharan Sahel region (Carrete et al., 2013; López-López et al., 2014).

The study was conducted from 2000 to 2013 in the Cádiz and Málaga regions of Andalusia (Southern Spain) (Fig. 1). Egyptian vultures are distributed around hills and mountain piedmonts. Four to six supplementary feeding stations have been available in the study area, which would indicate that food resources are not a limiting factor (Benítez et al., 2001; Margalida et al., 2012). During the study period, 175 Egyptian vultures were captured and individually marked with both darvic and aluminum rings: 159 fledglings were captured in their nests and 4 individuals aged 1 yr-old, 4 aged 2 yrs-old, 3 aged 3 yrs-old, 1 aged 4 yrs-old and 4 breeding adults (unknown age) were captured using cannon nets. Age was determined on the basis of plumage characteristics (Cramp and Simmons, 1977). Resightings of marked individuals during the breeding season ($n = 100$) and recoveries of dead birds ($n = 3$) were recorded until 2013.

Thirty-three breeding territories were found in the study area from 2000 to 2013. From 2000 to 2004, 20 to 32 territories were annually prospected, but from 2005 onwards monitoring efforts were intensified and all known territories (including active and abandoned territories) and all those cliffs *a priori* identified to be adequate to hold a breeding pair, were prospected every year. Territories were prospected every two weeks (typically spending 1 to 2 h per visit) during the breeding season and data on territory occupancy and breeding success (i.e., number of chicks fledged by territory and year) was collected. When adult absences were detected, nest sites were carefully inspected to detect potential dead birds inside nests. The territories adjacent to nest sites, foraging areas and nearby wind farms were then covered looking for carcasses (Carrete et al., 2009). Causes of death were determined by necropsy and pathology procedures (Hernández and Margalida, 2009).