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European policies on livestock carcasses management did not modify the foraging behavior of a threatened vulture



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ABSTRACT

Policy decisions have the potential to affect biodiversity conservation and modify, among other factors, animal demography, behavior and ecological processes. This became manifest in Europe in the past decade in light of the appearance of bovine spongiform encephalopathy (BSE). However, the obvious effects of policy decisions on foraging behavior are lacking. Here, we use data from a long-term study (1992–2015) of the ecology of the bearded vulture *Gypaetus barbatus* in the Pyrenees to assess whether or not the implementation of health policies in the period 2006–2011 really did affect this vulture's foraging behavior and breeding success. To test this, we used the frequency of nest changeovers (between 1992 and 2015) as a surrogate for the time invested in searching for and obtaining food. Additionally, between 2006 and 2015 we monitored the movements of 20 bearded vultures with GPS satellite transmitters. Our findings show no differences between annual home range sizes and/or the time used to search for food either before, during or after the imposition of health regulations. No differences between periods (during or after health regulations) were found in the use of supplementary feeding sites (SFS). An analysis of food available indicates that this similarity of use in different periods could be explained by the high habitat quality in the Pyrenees and the good availability of trophic resources. Our results suggest that diet plasticity and habitat quality compensate for the sudden effects of food shortages and so question the true usefulness of SFS in buffering a quantitative food deficit. We discuss the implications of these results from both ecological and conservation perspectives.

1. Introduction

An assessment of the effects of anthropogenic activity on ecosystem dynamics is often complex since these effects may not manifest themselves until several years after the original human impact has occurred (Vitousek et al., 1997; Robertson et al., 2012). In addition, the dearth of previous long-term datasets that can assess objectively the relationships between human actions and ecosystem effects hampers any adequate evaluation of this cause/effect relationship (Robertson et al., 2012); this lack thus highlights the need for long-term studies (Mills et al., 2015). This scenario becomes evident when sudden changes due to policy decisions affecting habitat quality (e.g. food availability) occur, a good example of which was the 2001 outbreak of bovine spongiform encephalopathy in Europe that brought out into the open contradictions between sanitary and environmental policies (Tella, 2001; Donazar et al., 2009a,b; Margalida et al., 2010). In Spain, home to > 90% of all European vulture species' individuals, the feeding

resources of both facultative and obligate scavenger species were suddenly limited by the application of health policies that, as a precautionary measure, obliged the carcasses of domestic animals to be collected and then transformed or destroyed in authorized plants, thereby reducing substantially (around 80%) the food available for obligate and facultative scavenger species (Donazar et al., 2009a,b; Cortés-Avizanda et al., 2010). Furthermore, the closure of several supplementary feeding sites (hereafter SFS) or 'vulture restaurants' accentuated the sudden changes occurring in the availability of trophic resources (Donazar et al., 2010; Blanco, 2014; Llana and López-Bao, 2015).

The new laws were implemented from 2002 onwards with the entry into force of Regulation EC 1774/2002 and were strongly enforced in Spain in, above all, 2005–2011. Nevertheless, since 2012 new regulations have made sanitary policies more flexible and have partially reversed the previous situation (EC 142/2011, Royal Decree 1632/2011; Margalida et al., 2012) by allowing farmers to leave extensive

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livestock carcasses in specific areas (*Protected Areas for the Feeding of Scavengers* – PAFs- implemented since 2013–2014; see Morales-Reyes et al., 2017) and/or feeding sites (Margalida et al., 2012).

The detrimental effects of the initial policies regarding scavenger species were highlighted by several studies that described shifts in dietary habits (Donazar et al., 2010; Blanco 2014; Llaneza and López-Bao, 2015), changes in behavioural ecology (Zuberogoitia et al., 2010; Margalida et al., 2011) and a fall in demographic parameters (Margalida et al., 2014). More recently it have been also documented additional detrimental effects related with the new regulations, as those affecting the greenhouse-gas emissions associated with the transport of carcasses (Morales-Reyes et al., 2015). However, it is still unknown how these policies will affect foraging behaviour, a serious shortfall given the importance of this information from management and conservation perspectives. In addition, Spain have an important network of supplementary feeding sites (Moreno-Opo et al., 2015; Cortés-Avizanda et al., 2016), being a good scenario for evaluating whether an intensification in the use of supplementary feeding sites occurred after food shortages and whether this affected the feeding and foraging movements of obligate scavengers (Deygout et al., 2009; Montsarrat et al., 2013).

Foraging occupies a substantial amount of the daily time budget of avian scavengers. The spatial and temporal unpredictability of food resources obliges individuals to explore extensive surface areas and optimal foraging behaviour is influenced by the quality of the foraging patch and the fitness of the individual (Houston and McNamara, 1986; McNamara and Houston, 1986). This uncertainty provokes a trade-off between energy gains and survival (Olsson et al., 2002). Limitations in food supplies may force animals to evaluate trade-offs between survival and fitness (Olsson et al., 2002) and so food availability is an important driver of home foraging ranges (DeVault et al., 2004; Carrete and Donazar, 2005; Kertson and Marzluff, 2011). The potential effects of food shortages on the use of space by vultures have not yet been determined but could be useful for managers and policy-makers. For example, changes in the use of space could determine just how important SFS are as tools for compensating a reduction in food supplies and/or how these effects are linked to demographic parameters (see Margalida et al., 2014). Nevertheless, the ways in which food shortages affect foraging and act as a key driver of animal space use have not been resolved. Resource distribution and temporal availability affects movement ecology but also individual life history and has a demographic impact on population dynamics (Nathan et al., 2008). According to ecological theory, birds may need to use a greater variety of food resources when the abundance of their preferred prey becomes scarcer (Bell and Ford, 1990).

We used data from a long-term study of the breeding ecology of bearded vultures that began in 1992. Thus, we had at our disposal information on nest changeovers and foraging behaviour that coincided with the food-shortage period, i.e. the Policy-Implementation Period (PI) in 2005–2011, as well as the pre- and post- food-shortage period, i.e. the No-Policy-Implementation Period (NPI) that encompasses the rest of the study period (1992–2004 and 2012–2015).

We hypothesized that if food shortages affect habitat quality and by extension breeding behaviour, foraging movements would vary according to the availability of resources and could, additionally, affect breeding parameters. Thus, hypothesis 1 was that health policies regulate food availability and affect home range size and daily movements, which means that home ranges and daily movements are larger when food availability is lower (during PI). This implies that the time invested in searching for food could affect breeding output. The species would respond to changes in food availability by increasing its consumption of alternative prey items and/or visiting SFS more frequently, albeit without modifying substantially the area it exploited and the time invested in finding food. Thus, a second hypothesis (hypothesis 2) was that SFS can buffer the effects of food shortages as individuals could use these predictable sites more frequently during PI and, consequently, modify their feeding and foraging patterns asso-

ciated with these points. As a result, a reduction in food availability could lead to greater use of SFS during PI, because these sites provide predictable food resources in a spatial and temporal scale. In addition, food shortages also could provoke differences between the PI and NPI periods in the presence of individuals at these sites — but without necessarily affecting the amount of time invested in searching for food during the incubation or chick-rearing periods or having any effect on breeding output.

To test this, we used three indicators 1) the frequency of changeovers during incubation and chick-rearing as a surrogate of the time invested searching for food. Thus, comparing the frequency of changeovers between PI (2005–11) and NPI (1992–1994 and 2012–2015) we can assess whether or not the foraging time varied as a response to the implementation of health policies; 2) the kernel and minimum convex polygon (MCP) estimators of home ranges to assess if there were differences between the PI (2006–2011) and NPI (2012–2015) periods in terms of the use of space, i.e. the foraging areas visited; and 3) the percentage of locations near SFS as a surrogate of a higher/lower use of these predictable food-resource sites in the PI (2006–2011) and NPI (2012–2015) periods. Additionally, we assessed the potential effects of changes in food availability on breeding output during this period by comparing productivity and breeding success during the PI and NPI periods.

2. Material and methods

2.1. Study area and data collection

The southern side of the Pyrenees (SW Europe) has been intensively monitored as part of the recovery plan for this species executed in the Spanish Autonomous Communities of the Basque Country, Navarre, Aragon and Catalonia that includes programs to monitor population trends, breeding parameters and survival rates (including a specific capture-mark-resighting subprogram).

In 1992–2015, all known territories located in the eastern Pyrenees (range 15–44 territories) were visited (2–3 visits/month) to search for signs of occupancy (territorial and/or courtship activity, nest arrangement/building) and to record reproductive parameters. Observations began in September–October, coinciding with the start of nest-building and sexual activity (Bertran and Margalida, 1999), and ended during fledging (June–August).

2.2. Bearded vulture capture and tracking

In 2006–2013 a total of 20 bearded vultures were fitted with satellite transmitters. Birds were captured using radio-controlled bow-nets at feeding sites ($n = 16$) or as fledglings in nests ($n = 1$), or were individuals from official wildlife recovery centres that were subsequently released ($n = 3$). Birds were assigned to one of four age-classes as per previous studies (Margalida et al., 2016): juvenile (birds during their first year of life), immature (2–3 years), sub-adult (4–5 years) or adult (≥ 6 years). Individuals were sexed from blood samples, while territoriality, breeding status and annual breeding success were assessed by field observations.

Solar-powered 70-gr (GPS/PTT) satellite transmitters (Microwave Telemetry, Inc. Columbia, MD) were used to track movements. Transmitters were attached to a backpack harness made from 0.64-cm Teflon ribbon (Bally Ribbon Mills, Bally, PA) and were programmed to collect GPS locations (18 m manufacturer's estimated error) every hour from 6:00 to 21:00 UTC; the only exception were two non-territorial adults that were programmed to transmit a fix every two hours. The average number of fixes collected per day for all tracked birds was 5.1 ± 2.9 (range per individual was 3.6 ± 2.4 – 7.5 ± 3.0).

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