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CONSERVACIÓN Y GESTIÓN DEL HÁBITAT DE REPRODUCCIÓN Y DE ALIMENTACIÓN DEL BUITRE NEGRO *AEGYPIUS MONACHUS* (LINNAEUS, 1766)

CONSERVATION AND MANAGEMENT OF THE FORAGING AND BREEDING HABITAT OF THE CINEREOUS VULTURE AEGYPIUS MONACHUS (LINNAEUS, 1766)

TESIS DOCTORAL DE:

RUBÉN MORENO-OPO DÍAZ-MECO

BAJO LA DIRECCIÓN DE:

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Conservación y gestión del hábitat de reproducción y de alimentación del buitre negro *Aegypius monachus* (Linnaeus, 1766)

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Tesis Doctoral

Rubén Moreno-Opo Díaz-Meco

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RESUMEN

El estudio de las relaciones existentes entre distintos componentes de los ecosistemas permite, entre otros, conocer las causas que determinan la presencia de algunos taxones en determinados emplazamientos así como la preferencia de éstos por distintos elementos de los hábitats. Para el caso de especies o poblaciones que suscitan un mayor interés, bien por su estado desfavorable de conservación o bien por su papel clave en los ecosistemas, el reconocimiento de las características del hábitat seleccionadas resulta muy importante de cara a la identificación de posibles factores limitantes o de amenazas. Disponer de esta información es un paso previo al diseño y aplicación de medidas destinadas a favorecer a la especie: se incrementa la precisión en el diagnóstico de la situación, se comprenden mejor las líneas prioritarias de trabajo a implantar y, en definitiva, aumentan las probabilidades de éxito de las actuaciones de gestión proyectadas.

El buitre negro *Aegypius monachus* es la mayor de las aves rapaces europeas y ha sido objeto en España de numerosos trabajos de seguimiento, estudio y conservación, debido a que su población era muy escasa tras la realización de los primeros censos en las décadas de 1970 y 1980. Forma parte del gremio de las aves necrófagas ibéricas, cuyas poblaciones son las más abundantes de todo el Paleártico occidental. Estas aves necrófagas juegan un importante papel en los ecosistemas ibéricos; así, por ejemplo, prestan servicios derivados del procesado de los animales muertos de manera sostenible, higiénica y económica. Además, el buitre negro es una especie indicadora de determinadas características de los espacios naturales que habita y constituye un buen modelo de estudio para evaluar qué factores son más representativos en el mantenimiento de condiciones favorables para especies superpredadoras o necrófagas en los medios forestales y agrarios mediterráneos. También, de su estudio, resulta reveladora su relación con las actividades humanas, el impacto que éstas tienen y cómo pueden reconducirse y compatibilizarse. Las principales amenazas para la especie provienen de malas prácticas realizadas por el hombre, como son el envenenamiento intencionado o la alteración del medio natural.

En este escenario, los objetivos de la presente Tesis Doctoral son 1) profundizar en el conocimiento de las relaciones existentes entre el buitre negro y su hábitat, 2) determinar los factores y características del hábitat que resultan más determinantes para el asentamiento de nidos de la especie y para su éxito reproductivo, así como para la selección de determinadas áreas de alimentación y tipos de alimento, 3) evaluar el efecto de determinadas actividades humanas en áreas de reproducción y alimentación, 4) establecer patrones de gestión que eviten los efectos negativos de las actividades realizadas por el hombre sobre el buitre negro, y 5) analizar el estado de conservación de la especie y las posibles causas que han incidido en la tendencia poblacional registrada en los últimos años en España. Para la consecución de los mencionados objetivos se aplicaron distintas técnicas de estudio. Para los puntos 1 y 2 se planificó un muestreo de variables geomorfológicas, ecológicas y antrópicas con el fin de evaluar los factores que determinan tanto la selección de hábitat de reproducción como el éxito reproductivo. Además, para esclarecer patrones de selección del hábitat de cría a escala

global, se llevó a cabo un metanálisis basado en los resultados obtenidos en otros estudios previos, en el que también se evaluó el efecto de la aplicación de distintos procedimientos metodológicos. Por otro lado, se realizó un seguimiento detallado de aportes en los que se controlaron las características de los aportes y las del hábitat circundante. También se recopilaron sistemáticamente crotales de ganado, como elementos indicativos para conocer, por medio de sistemas de información geográfica, las áreas de campeo y de alimentación en torno a una colonia de cría de buitre negro. En relación a los objetivos 3 y 4 se evaluó la incidencia del trabajo de descorche de los alcornoques existentes en una colonia de cría sobre los buitres negros reproductores y sobre sus pollos. Además, se llevó a cabo un ensayo para conocer la aceptación tanto del buitre negro como de otras rapaces necrófagas a aportes de carroña depositados en vallados perimetrales móviles, dispuestos con la intención de evitar la entrada de otras especies necrófagas no objetivo y optimizar así el manejo de los puntos de alimentación suplementaria. Por último, para explorar el objetivo 5 se recopilaron publicaciones relacionadas con el buitre negro y, con la información obtenida, se realizó un análisis de las posibles relaciones causa-efecto entre el esfuerzo de investigación y de conservación y la tendencia poblacional de la especie en España.

Los trabajos de campo se desarrollaron entre los años 2003 a 2011 en España centro-occidental. De manera específica, los muestreos relativos al estudio de factores asociados a la reproducción y a la actividad de campeo se ejecutaron en la colonia de cría de la Umbría de Alcudia, Castilla-La Mancha, mientras que los estudios sobre preferencias de presa en carroñas o de evaluación de vallados en puntos de alimentación suplementaria se desarrollaron en distintas zonas de especial protección de aves (ZEPA) de Extremadura y Castilla-La Mancha.

Los resultados obtenidos muestran la importancia de masas forestales con árboles grandes y con elevada cobertura vegetal, alejadas de la actividad humana y en áreas de difícil acceso como enclaves seleccionados por el buitre negro para emplazar sus nidos. Además, valores elevados de estas mismas variables, junto con un mejor estado de conservación de las plataformas de nidificación, incrementan las posibilidades de éxito reproductivo. En relación a los métodos de estudio de la selección de hábitat a escala global, la aplicación de métodos diferentes de muestreo y análisis entre estudios determinó la variabilidad recogida en los modelos resultantes. Así, la elección de determinados procedimientos experimentales en cada estudio implicó una mayor varianza/devianza: puntos de muestreo > parcelas de muestreo, puntos aleatorios de muestreo dentro de la colonia > puntos aleatorios fuera de la colonia, un único año de estudio > más de un año de estudio, y mayor proporción de puntos aleatorios respecto nidos > menor proporción de puntos aleatorios respecto nidos. En definitiva, tanto la elección de los protocolos de muestreo como la homogeneización de éstos entre los distintos estudios es importante para comparar adecuadamente resultados de selección de requerimientos ecológicos en un contexto global.

El buitre negro resulta más abundante en los aportes de carroña cuando ésta se dispone dispersa en una superficie amplia y en elevadas cantidades. De esta manera optimiza la adquisición de alimento respecto al buitre leonado *Gyps fulvus*, principal especie competidora, sobre todo cuando la carroña se presenta concentrada en un único punto y en forma de cadáveres enteros. Además, el buitre negro consume preferentemente trozos dispersos pequeños y medianos, de músculos y tendones. Promover esta tipología de aportes cuando se realizan actuaciones de alimentación suplementaria o cuando se depositan despojos de ungulados cazados resulta muy favorable para el buitre negro y para otras necrófagas amenazadas como el alimoche común *Neophron percnopterus* o el milano real *Milvus milvus*. Durante el período de reproducción los adultos acuden a los aportes de alimento en mayor medida, mientras que la proporción de jóvenes y subadultos en las carroñas permanece constante durante todas las fases del ciclo vital. Respecto a las áreas de campeo en áreas de aprovechamiento ganadero, la localización de los crotales mostró la presencia de ejemplares alimentándose en un total de 66,732.28 km^2 (Máximo Polígono Convexo, MCP) y, en base a una probabilidad de presencia del 95% (análisis Kernel), en una superficie de 152.290,13 ha. El promedio de los desplazamientos en línea recta realizados por los ejemplares desde el punto de alimentación a la colonia de cría fue de 26.3 km. Con este trabajo, se ha revelado un sistema novedoso de estudio de áreas de campeo que presenta ciertas limitaciones y divergencias metodológicas respecto a las técnicas empleadas en otros estudios análogos.

Las actividades de aprovechamiento forestal en áreas de cría tienen repercusión negativa sobre el éxito reproductivo del buitre negro si no se realizan de manera adecuada. En concreto, el estudio del efecto del descorche puso de manifiesto que los adultos abandonaban el nido cuando se rebasaba por parte de los trabajadores una distancia promedio de 132 m, sobre todo cuando el ruido producido era alto o medio. Si en esas circunstancias el pollo era menor de 40 días de edad, el abandono se producía en las horas de más calor y la molestia se repetía en días consecutivos, la probabilidad de fracaso se incrementaba. Así, en áreas sometidas a descorche el éxito reproductivo se redujo un 20% y la mortalidad de pollos en nido se triplicó. No obstante, el descorche es una actividad tradicional y necesaria para el mantenimiento de los alcornocales donde habita el buitre negro y se produce cada nueve años. Por ello, es preciso establecer mecanismos de trabajo dirigidos a reducir al mínimo las molestias, adecuando algunos patrones de actuación sobre horarios, zonas de trabajo y reducción del nivel de ruido. En relación a la gestión de la alimentación de especies necrófagas, resulta necesario satisfacer los requisitos sanitarios que disminuyan el riesgo de transmisión de enfermedades, al tiempo que las especies necrófagas satisfacen sus requerimientos tróficos en relación a su disponibilidad, aparición en el territorio y calidad. Para cumplir los anteriores requisitos puede resultar de ayuda la implantación de vallados perimetrales móviles que impidan la entrada a la carroña de especies no objetivo que podrían actuar como vectores de enfermedades transmisibles. Las mallas electrificadas móviles resultan eficaces en este sentido y no alteran el acceso natural de aves necrófagas a la carroña. Su fácil montaje y desmontaje las convierten en un sistema cómodo y resolutivo para poder alternar distintas áreas de aporte de alimento dentro de una misma explotación.

Por último, y como conclusión de la Tesis, del análisis de las distintas publicaciones sobre el buitre negro se concluye que la población de la especie se ha podido ver favorecida por el incremento del conocimiento sobre aspectos básicos de su ecología y conservación y por la publicación de resultados en revistas científicas de impacto. La

protección de la propia especie y la de sus colonias de cría supusieron un hito importante entre las décadas de 1970 a 1990. La tendencia positiva de la población está correlacionada con la de una de sus fuentes de alimento más importantes, los ungulados silvestres. En este sentido, de la adquisición del alimento provienen los principales factores de riesgo, como son la escasez de determinadas presas como el conejo de monte *Oryctolagus cuniculus*, la disminución de cadáveres de origen ganadero por las políticas de gestión de subproductos iniciadas en la década de 2000 o la ingestión de cebos envenenados. A la luz de la recopilación de información, los trabajos de conservación de buitre negro han propiciado una evolución positiva de su contingente poblacional en España, a pesar de que en las primeras fases no se contaba con un conocimiento tan adecuado como el que se dispone actualmente.

ABSTRACT

The study of the relationships between ecosystem components allows, among other things, a greater knowledge of factors determining the presence of some taxa in certain locations, and the preference of these taxa for different habitat characteristics. In the case of species or populations of great interest, either due to their unfavorable conservation status or because of their key role in ecosystems, recognition of the selected habitat characteristics is very important in order to identify potential limiting factors or threats. Acquiring this information is a prerequisite for the design and implementation of conservation measures: it enhances the accuracy of diagnosiing the problem, a better understanding of the priority work to be implemented and, ultimately, it increases the likelihood of success of planned management actions.

The cinereous vulture *Aegypius monachus* is the largest European raptor and has been the subject of a great number of monitoring, research and conservation programs in Spain, since its population was shown to be very low in the first censuses in the 1970s and 1980s. It belongs to the guild of Iberian necrophagous birds, whose populations are the most abundant of the Western Palearctic. These scavengers play an important role in Iberian ecosystems by, for example, processing dead animals in a sustainable, hygienic and economic way. In addition, the cinereous vulture is an indicator of certain characteristics of the environment and is a good case study to assess the factors that are most representative and determinant in maintaining favorable conditions for scavenger and top-predator species within Mediterranean forests and agricultural lands. Moreover, from the study of this species we can learn about its relationship to human activities, the impact these activities have on population dynamics and how these activities can be redirected and compatibilized to avoid negative effects. The main threats to the species come from human activity, including poisoning or alteration of the natural habitats.

Given this scenario, the objectives of this Thesis are 1) to achieve a deeper understanding of the relationship between the cinereous vulture and its habitat; 2) to identify the most crucial factors for the settlement of nests of the species, for its breeding success and for the selection of feeding areas and types of food; 3) to evaluate the effects of certain human activities on breeding and foraging areas; 4) to determine management measures that mitigate the negative effects of human activities on the cinereous vulture; and 5) to assess the conservation status of the species and the causes of the recent population trends in in Spain. To achieve these objectives different study techniques were applied. For goals 1 and 2 geomorphological, ecological and anthropogenic variables were sampled in order to evaluate the factors that determine both the selection of breeding habitat and breeding success. Moreover, to clarify patterns of breeding habitat selection at a global scale, a meta-analysis based on the results of previous studies was conducted, which also evaluated the effect of applying different methodological procedures in the achieved results. On the other hand, a survey of feeding events of the cinereous vulture was performed in which the characteristics of the inputs and the surrounding habitat were monitored. We also systematically collected livestock ear tags as indicative of feeding locations of cinereous vultures to reveal, through GIS techniques, foraging and feeding areas around a breeding colony. Regarding goals 3 and 4 we evaluated the incidence of cork exploitation in a breeding colony, assessing its effect on breeding cinereous vultures and their chicks. In addition, a study was performed to determine the whether the cinereous vulture and other scavenger raptors accepted carrion inputs provided within mobile perimeter fences. These exclosures were installed with the intention of preventing the entry of non-target scavenger species and were aimed at optimizing the management of supplementary feeding sites. Finally, to explore goal 5, publications related to the cinereous vulture were compiled with the aim of comparing and quantifying possible cause-effect relationships between research effort and population trends of the species in Spain.

Fieldwork took place between 2003 and 2011 in west-central Spain. Specifically, the study of the factors associated with breeding and foraging activities was implemented in the Umbria de Alcudia breeding colony, Castilla-La Mancha, while studies on prey preferences and evaluation of different supplementary feeding procedures were developed in different special protection areas for birds (SPAs) of Extremadura and Castilla-La Mancha regions.

The results showed the importance of forests with large trees and well-developed scrub cover, far from human activities and in areas of difficult accessibility, for selecting the location of the nests. In addition, high values for these variables, along with a better conservation status of nesting platforms, increase the chances of breeding success of the species. In relation to habitat selection at different locations, the application of different sampling and analysis methods across studies determined the variability contained in the models. Thus, the choice of certain experimental procedures involved greater variance / deviance values within the results: sampling points> sampling polygons, random points within the colony> random points outside the colony, a single year of study> more than one year of study, and a higher proportion of random points in relation to nests> lower proportion of random points in relation to nests. In short, both the choice of sampling protocols and the homogenization of protocols between different studies are important for a proper comparison of species requirements in a global context.

In relation to feeding, the cinereous vulture was more abundant when carrion was spread over a large area and in high quantities. These conditions optimize the ingestion of food compared to Eurasian griffon vultures *Gyps fulvus*, the most competitive species, especially when food is concentrated at single points and as entire carcasses. In addition, the cinereous vulture selects scattered, small and medium pieces of muscles and tendons. Promotion of these characteristics when providing carrion in supplementary feeding programs can be highly favorable to the cinereous vulture and to other threatened raptors such as the Egyptian vulture *Neophron percnopterus* or the red kite *Milvus milvus*. During the breeding period adults attend to inputs of carrion in higher numbers, while the proportion of young and subadult birds at carrion inputs remains stable throughout the breeding cycle. Regarding the home ranges of the cinereous vulture, the location of the identified livestock tags showed a foraging range of 66,732.28 km^2 (maximum convex polygon, MCP) and, based on a likelihood of presence of 95 % (kernel analysis), of 152,290.13 ha. The mean distance of flights made by individuals from the breeding colony to the feeding point was 26.3 km. As a result of this work, a novel system for studying foraging areas was implemented, with some methodological differences from techniques used in other similar studies.

Logging activities within cinereous vulture breeding areas have negative impacts on breeding success. Specifically, the study of cork exploitation showed that adults left the nest when workers came closer than an average distance of 132 m to the nest, especially when the noise level was high or at mid-level. If under these circumstances the chick was under 40 days of age, the activity occurred during the hottest hours of the day and the presence of workers occurred on consecutive days, the probability of breeding failure increased. Thus, in areas subject to cork exploitation breeding success decreased by 20% and the mortality of chicks in nests tripled. On the other hand, cork exploitation is traditional and neccessary for the conservation of cork oak forests inhabited by the cinereous vulture and harvesting occurs only every nine years. It is therefore necessary to establish protocols aimed at minimizing disturbances by adjusting some working patterns to defined periods and areas and by reducing the level of noise made by the workers. Regarding the management of supplementary feeding points for avian scavengers, it is necessary to satisfy sanitary conditions that reduce the risk of disease transmission while scavengers meet their feeding requirements regarding availability, quality and occurrence. To meet the above conditions when providing carrion, the implementation of mobile, affordable and easy to install/dismantle perimeter fences may help to prevent entry by non-target species that could act as vectors of disease. The mobile electrified mesh fences are effective in this regard and do not alter natural access to the food of avian scavengers. Its easy assembly and disassembly makes it a reasonable option for alternating feeding points for scavengers within the same estate or farm.

Finally, in conclusion of this PhD Thesis, the analysis of publications on the cinereous vulture showed that the Spanish population could benefit by increased knowledge about basic aspects of its ecology and conservation status published in journals of scientific impact. The protection of the species and breeding colonies represented a major milestone in the 1970s to 1990s. The positive trend of the population is correlated with one of its most important food sources, wild ungulates. Otherwise, the major risk factor for the species is lack of appropriate food sources, due in part to the scarcity of preys such as the wild rabbit *Oryctolagus cuniculus*, the alteration in availability and occurrence of livestock carcasses as a result of the European management policies for animal by-products and the ingestion of poisoned baits. In light of the information collected, the initiatives and projects for cinereous vulture conservation have led to an increase in the Spanish population.

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Todo empezó con la planificación del proyecto LIFE *Conservación del águila imperial ibérica, el buitre negro y cigüeña negr*a, allá por el año 2003. En la Fundación CBD-Habitat pudimos pergeñar varios protocolos de seguimiento con el objetivo de informar sobre la eficacia de las medidas de actuación que contemplaba el proyecto. Gracias a este trabajo de seguimiento se obtuvieron valiosísimos datos de campo que conforman la mayor parte de la base de conocimiento que integra esta Tesis Doctoral. Resultó clave la confianza de Javier Oria y Nuria El Khadir, como jefes, y la coordinación y trabajo conjunto con Francisco Guil (Curro), con el que ha sido un placer compartir tantos momentos. El trabajo de campo recayó en algunos de los mejores profesionales con los que he tenido la suerte de compartir vivencias y con los que he llegado a consolidar, en muchos casos, una amistad inquebrantable: Ángel Arredondo, Rafael Higuero, Manuel Martín, Carlos Soria, José Guzmán, Rosa Jiménez, Maxi Panadero y Samuel Pla. Fuimos un grupo buenísimo y lo recuerdo de forma muy grata. El resto de compañeros de la Fundación CBD-Habitat siempre apoyó en lo posible y forman (o formaron) un equipo humano y técnico maravilloso: la gente del lince (Fernando Silvestre, Sandra Agudín, Javier Inogés, Juan Francisco Sánchez, Manuel Mata, Juan Figueredo), los de la foca monje (Michel Cedenilla, Pablo Fernández, Hamdi M´Barek, Moulaye Haye, Ana Maroto, Mercedes Muñoz) y la "transversal" de la oficina (Lourdes Ortega, Cristina Martínez). El trabajo de campo fue posible gracias a la colaboración prestada por guardas y propietarios de las fincas privadas donde se desarrolló el LIFE, en algunas de las zonas de monte mediterráneo mejor conservadas. Compartir inquietudes, conocimientos y planteamientos con ellos me acercó a la realidad del campo. Esta interacción es primordial para que cualquier iniciativa que se plantee "desde despachos" pueda tener éxito en la práctica.

En el desarrollo de la presente Tesis Doctoral resultaron clave distintas personas que me animaron, asesoraron y ayudaron de manera fundamental. Compartir trabajo con Antoni Margalida fue un regalo del destino. Poder recibir consejos, plantearle dudas y aprender a publicar artículos resultó (y espero continúe en el futuro) un vicio sano y raro, de esos que surgen aunque a uno no le paguen por ello, dedique tiempo de donde no lo hay (bueno sí, sobre todo de la familia) y haga esforzarte día a día. Él me animó a enfocar los distintos trabajos hacia una Tesis Doctoral, tal y como él realizó brillantemente con la suya sobre el quebrantahuesos, lo que constituyó el mejor ejemplo. Posteriormente me atreví a pedir a José Luis Tellería que me dirigiera la Tesis Doctoral, tras haber realizado el Máster en Biología de la Conservación en la Universidad Complutense de Madrid. Y tuve el lujo de que aceptara, de modo que soy de los pocos afortunados que ha podido escucharle, durante más tiempo del que duran las clases, hablar de pájaros, de conservación, de métodos de muestreo y de tantísimas cosas de las que tanto sabe. Le admiro profundamente por su manera de enfocar el

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LISTA DE PUBLICACIONES ORIGINALES

Esta Tesis Doctoral está basada en siete artículos científicos que conforman el contenido de los siete capítulos, y dos publicaciones divulgativas que contribuyen a la elaboración de parte de la introducción.

Introducción

García de Francisco, J. M. and Moreno-Opo, R. (2009) Livestock carcass management today: is there enough flexibility to deal with new conservation strategies? In: Donázar, J. A., Margalida, A. & Campión, D. (Eds.) Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology. *Munibe* 29. Sociedad de Ciencias Aranzadi. San Sebastián, pp. 502-509.

Moreno-Opo, R. and Guil, F. (2007) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

Capítulo 1

Moreno-Opo, R., Fernández-Olalla, M., Margalida, A., Arredondo, A. and Guil, F. (2012) Effect of methodological and ecological approaches on heterogeneity of nest-site selection of a long-lived vulture. *PLoS ONE* 7: e33469.

Capítulo 2

Moreno-Opo, R., Fernández-Olalla, M., Margalida, A., Arredondo, A. and Guil, F. (*En revisión*) Influence of environmental factors on cinereous vulture *Aegypius monachus* breeding success in central Spain.

Capítulo 3

Moreno-Opo, R., Arredondo, A. and Guil, F. (2010) Foraging range and diet of cinereous vulture *Aegypius monachus* using livestock resources in central Spain. *Ardeola* 57: 111-119.

Capítulo 4

Moreno-Opo, R., Margalida, A., Arredondo, A., Guil, F., Martín, M., Higuero, R., Soria, C. and Guzmán, J. 2010. Factors influencing the presence of cinereous vulture *Aegypius monachus* at carcasses: food preferences and implications for the management of supplementary feeding sites. *Wildlife Biology* 16: 25-34.

Capítulo 5

Margalida, A., Moreno-Opo, R., Arroyo, B. E. and Arredondo, A. (2011) Reconciling the conservation of endangered species with economically important anthropogenic activities: interactions between cork exploitation and the cinereous vulture *Aegypius monachus* in Spain. *Animal Conservation* 14: 167-174.

Capítulo 6

Moreno-Opo, R., Margalida, A., García, F., Arredondo, A., Rodríguez-Vigal, C. and González, L. M. (2012) Linking sanitary and ecological requirements in the management of avian scavengers: effectiveness of fencing against mammals in supplementary feeding sites. *Biodiversity and Conservation* 21: 1673-1685.

Capítulo 7

Moreno-Opo, R. and Margalida, A. (*En revision*) Conservation of the cinereous vulture *Aegypius monachus* in Spain (1966-2011): threats, research and adaptive management.

INTRODUCCIÓN

Conservación de especies amenazadas

Sobre la Biología de la Conservación

A lo largo del siglo XX y primeros años del siglo XXI se ha producido una alteración en la dinámica de los sistemas naturales a escala global que ha generado una alerta, avisada por expertos y organizaciones científicas, sobre las posibles consecuencias negativas para la sostenibilidad del planeta (Wilson 1992, Rockström et al. 2009). Estas alteraciones, agrupadas bajo el concepto de cambio global, son consecuencia de la modificación de distintos parámetros físicos, químicos y ecológicos, como la temperatura media, la dinámica de precipitaciones, la superficie ocupada por bosques maduros o el aumento de la concentración de dióxido de carbono en la atmósfera (Vitousek 1994, Sala et al. 2000). Una consecuencia de esta situación ha sido la aceleración de la tasa de extinción de especies, que se ha multiplicado por entre 100 y 1.000 veces (Brooke et al. 2008, Rockström et al. 2009). De hecho, el ritmo de extinción supondrá, de no remediarse, la desaparición de cientos de especies en los próximos 100 años y generará efectos perjudiciales para el hombre, económicos, sanitarios o estéticos, por la pérdida de funcionalidad de los ecosistemas (Sala et al. 2000, Hassan et al. 2005, Díaz et al. 2006).

En este escenario, surgió la Biología de la Conservación como disciplina científica que se encarga de estudiar las causas de la pérdida de diversidad biológica en todos sus niveles (genético, individual, específico y ecosistémico) y de proteger y restaurar dicha diversidad biológica a través de la búsqueda de soluciones que tengan en cuenta los principios básicos en los que se fundamenta: 1) la necesidad de conocer la distribución de la vida en el planeta y de las amenazas a las que ésta se enfrenta, 2) la evolución como mecanismo de generación de biodiversidad y de resolución de los problemas de conservación, 3) el dinamismo de los procesos ecológicos y la existencia de numerosos factores que lo afectan, y 4) la influencia de las actividades humanas en los sistemas ecológicos y la necesidad de considerar dichos efectos en la planificación de actuaciones (Soulé 1985, Meffe y Carroll 1994, Pullin 2002, Primack 2012).

Uno de los ámbitos abordados por la Biología de la Conservación es la protección de especies y/o poblaciones que se encuentran en un estado desfavorable de conservación y cuya probabilidad de extinción es elevada (Soulé 1985). La acción del hombre ha provocado modificaciones en el hábitat, en la dinámica de poblaciones y en el comportamiento de la mayoría de especies cuyo riesgo de extinción es alto, de modo que es preciso plantear iniciativas que reviertan sus tendencias demográficas y mejoren su estado de conservación (Novacek y Cleland 2001). Pero ¿es realmente necesario proteger las especies?, ¿por qué hay que invertir recursos en conservarlas? Existen varios principios que argumentan dicha necesidad: ético – la vida en sí misma es un acontecimiento inaudito y valioso puesto que cada especie ha sido originada a lo largo de millones de años de casualidades probabilísticas (Minteer y Collings 2005)-; estético – la consideración de que *la vida es bella* se debe a la existencia de todos y cada uno de los componentes que la forman-; científico – cada especie encierra numerosos enigmas aún por resolver para la ciencia que podrían ayudar a comprender mejor la vida en el futuro (Primack 2012)-; económico – todas las especies tienen un valor económico y suponen un recurso que beneficia al bienestar humano a través, por ejemplo, de alimentos, medicamentos, materiales, etc. (Carpenter et al. 2009, Primack 2012)-; complementariedad – las especies no están aisladas y forman parte de un entramado interrelacionado entre sí, en el que unas dependen de otras, de modo que la pérdida de una de ellas puede acarrear consecuencias negativas en cascada para el resto del ecosistema y, en consecuencia, su funcionalidad y valor económico (Soulé et al. 2005, Bascompte y Stouffer 2011, Estes et al. 2011)-.

Riesgo de extinción, listas de especies amenazadas y protección legal

Para evaluar el estado de conservación de especies y poblaciones se han usado distintos mecanismos en los últimos años. Todos ellos tienen como objetivo analizar el riesgo de extinción de los taxones, y uno de los más extendidos y consensuados es el propuesto por la Unión Internacional de Conservación de la Naturaleza (IUCN 2001), en base a criterios de selección y categorías de amenaza. La asignación de un nivel de amenaza depende de la información disponible acerca de la situación y tendencia de la población, del número de individuos o de la superficie/extensión de territorio ocupado. La información detallada sobre estas cuestiones es escasa para la gran mayoría de especies por lo que estimas precisas de riesgo pueden ser realizadas solo para unas pocas o, si se realizan, es en base datos poco robustos y concluyentes (O´Grady et al. 2004, Mace et al. 2008). De no contar con información fiable sobre tendencias de censos y distribución, otra opción recomendable puede ser el análisis del riesgo de extinción a través de técnicas que modelizan las poblaciones en base a rasgos biológicos y demográficos propios de la especie objetivo (población, tendencia, natalidad, mortalidad, reproducción, etc.), mediante los análisis de viabilidad poblacional PVA (Shaffer 1990, Boyce 1992). Se trata de una técnica que informa sobre las probabilidades de supervivencia de poblaciones y que, bajo determinados escenarios y supuestos, permite evaluar distintas prioridades de conservación (Carrete et al. 2009a, Ortega et al. 2009, Fernández-Olalla 2012). Desarrollos más avanzados de esta técnica permiten incorporar a los análisis aspectos genéticos y de viabilidad de hábitat (Schadt et al. 2002, Krame-Schadt et al. 2005) o de disponibilidad de recursos tróficos (Margalida et al. 2011a, Margalida y Colomer 2012), integrando un mayor abanico de circunstancias para la toma de decisiones sobre conservación.

La evaluación del riesgo de extinción y la definición de prioridades de conservación son dos ejercicios diferentes pero relacionados. La evaluación del riesgo de extinción, por lo general, precede a la definición de prioridades de actuación. Los criterios de categorización en listas rojas y libros rojos tienen como finalidad la estimación relativa de la probabilidad de extinción de un taxón. Por otro lado, los catálogos de especies amenazadas tienen como objetivo el establecimiento de prioridades de conservación, que llevan asociadas actuaciones de gestión con un refrendo legal y vinculante. La definición de prioridades de los catálogos oficiales toman en consideración el riesgo de extinción, pero también otros factores como su valor científico, ecológico y cultural, su singularidad y rareza, la interacción con las actividades tradicionales e históricas realizadas por el hombre, así como la probabilidad de éxito de las acciones de conservación, la disponibilidad de recursos económicos y/o humanos para ejecutar las actividades de conservación y los marcos legales para conservar las especies amenazadas (IUCN 2003).

Conservación de procesos ecológicos y evolutivos

Además de la protección de especies amenazadas, otros postulados derivados de la Biología de la Conservación abogan por la conservación de los procesos ecológicos y evolutivos que ocurren en los distintos ecosistemas, como mecanismo para restaurar la diversidad biológica de una manera más duradera, compleja y funcional (Brooks et al.

2004, Klein et al. 2009, Gilman et al. 2011). Dentro del estudio de los procesos ecológicos se integran otras disciplinas que lo desarrollan y complementan, como son la ecología del paisaje –composición y características-, la estructura de las comunidades ecológicas o los servicios ecosistémicos (Lindenmayer et al. 2008, Fisher et al. 2009, Tscharntke et al. 2012). La conservación de especies y de procesos ecológicos son complementarios aunque en determinadas ocasiones han surgido discusiones sobre la prioridad entre ambos tipos de actuación y cuál de ellos tiene una repercusión mayor para la conservación de la diversidad biológica (i.e. Franklin 1993, Mace et al. 2003, Likens y Lindenmayer 2012). En este sentido, el mantenimiento de los procesos ecológicos, entendidos como la matriz de relaciones entre los componentes de un ecosistema y que pueden incluir, por ejemplo, tasas de intercambio físico y químico entre elementos bióticos y abióticos, interacciones ecológicas, relaciones de competencia, etc., no ha sido incorporado como objeto de conservación hasta tiempos recientes (Bowker et al. 2008, Klein et al. 2009), debido principalmente a la dificultad de definirlos explícitamente en una escala espacial considerada (Lindenmayer et al. 2008). Ello ha hecho que los responsables políticos y gestores en conservación no lo hayan considerado tan frecuentemente como a las especies o los hábitats como criterios de elegibilidad en la declaración de espacios protegidos (e.g. European Commission 1992, Zisenis 2009). No obstante, esta tendencia está cambiando y, en la medida en que avanzan las técnicas para la definición y cuantificación de dichos procesos, se están integrando en planes y programas de conservación (Klein et al. 2009, Likens y Lindenmayer 2012).

La conservación de los procesos evolutivos se refiere a la preservación de la diversidad genética dentro de las poblaciones de cada especie. Esta diversidad genética intraespecífica puede deberse tanto a fenómenos históricos de aislamiento geográfico entre individuos como a mecanismos de evolución adaptativa debidos a presiones selectivas divergentes y que se manifiestan fenotípicamente en los individuos (Crandall et al. 2000, Moritz 2002). Ambas cuestiones han sido tratadas hasta tiempos recientes a nivel taxonómico, integrándose en subespecies, razas, etc., aunque dicha delimitación presenta limitaciones y subjetividades por la heterogeneidad en los criterios de discriminación, por el esfuerzo de estudio dedicado a las diferentes especies y por su falta de reflejo en evidencias moleculares y genéticas (Zink 2004). Para paliarlo, las Unidades Evolutivas Significativas (ESU, en inglés) son poblaciones consideradas singulares e independientes y conforman unidades prioritarias con fines de conservación (Moritz 1994). Estas ESU han evolucionado independientes de otras poblaciones y están delimitadas por características genéticas comunes (Crandall et al. 2000). La identificación de ESUs ayuda en el proceso de decidir qué poblaciones, dentro de cada especie, deben ser priorizadas para su conservación (Moritz 2002). Este concepto es complementario y coherente con la conservación de especies amenazadas, resultando un criterio más avanzado para determinar qué porción de la población de la especie es prioritaria para la dedicación de esfuerzos por sus características evolutivas.

Las especies amenazadas como motor de actuaciones de conservación

La protección de las especies más amenazadas se ha convertido en las últimas décadas en una de las acciones más importantes de conservación de la biodiversidad a escala global, junto con la declaración y gestión de espacios naturales protegidos (Margules y Pressey 2000, Primack 2012). Tras la progresiva aparición de la conciencia ecológica, surgió la necesidad de conocer de manera detallada los componentes de la vida silvestre, sobre todo en los países desarrollados. Así, surgieron programas de seguimiento y evaluación del estado de conservación de especies y hábitats (Baillie 1990, Magurran et al. 2010) y, como consecuencia de sus resultados, se ha podido conocer qué especies están en un estado de conservación más desfavorable. Gracias a estos programas y a otras actuaciones dirigidas a especies representativas, se han priorizado las especies sobre las que actuar (e. g. Maxwell y Jennings 2005, SEO/BirdLife 2010). La inversión económica realizada en los últimos años para salvar especies amenazadas ha tenido resultados variables (Ferraro y Pattanayak 2006). Algunas especies han logrado soslayar su estado crítico gracias a la ayuda proporcionada por el hombre (i.e. González et al. 2008, Simón et al. 2012), mientras que otras aún continúan amenazadas o se incrementa su riesgo de extinción (i.e. Hoffman et al. 2010, Waltson et al. 2010).

La dotación de recursos económicos a la protección de especies amenazadas ha permitido que, además de acciones destinadas a los propios individuos, se inviertan esfuerzos en proteger el hábitat y los espacios en los que dichas especies viven (Soulé et al. 2005). Existen actuaciones dentro de programas de conservación cuyos receptores son los propios individuos, como son las tareas de seguimiento, las acciones de conservación ex situ, el suministro de alimento o la sensibilización ambiental (Redford 2011). Pero por otro lado, gran parte de las propuestas emanadas de proyectos de conservación de especies se dirigen a restaurar y proteger los espacios habitados o potencialmente habitables. Así, actuaciones como mejoras de hábitat, eliminación de factores de mortalidad no natural, protección de áreas críticas y de períodos sensibles, declaración de espacios protegidos, compatibilización de usos económicos, resolución de conflictos con intereses humanos o vigilancia ambiental generan un efecto positivo sobre otros elementos de los ecosistemas, de forma genérica y a distintas escalas espaciales (Soulé et al. 2003, Redford et al. 2011, Maes et al. 2012). De este modo, las especies amenazadas receptoras de acciones de conservación que benefician a otras especies y elementos del medio natural pueden considerarse paraguas, según su rol y conexiones ecológicas con otros elementos del ecosistema (Lambeck 1997, Meffe et al. 2002, Larsen et al. 2007). La utilidad y rigor del concepto de especie paraguas ha sido discutido en numerosas ocasiones, destacando la ausencia de evidencias sobre repercusiones positivas generales sobre el resto de seres vivos, debido a las complejas relaciones y equilibrios existentes en cada una de las áreas y ecosistemas de que se trate (Seddon y Leach 2008, Branton y Richardson 2011, Lindenmayer y Likens 2011). No obstante, en tanto las actuaciones promovidas y ejecutadas para una especie amenazada puedan beneficiar a otras especies de su entorno, el concepto de especie paraguas puede contribuir a los fines últimos perseguidos en Biología de la Conservación (Suter et al. 2002, Roberge y Angelstam 2004). Un ejemplo de ello es la consideración de la presencia de especies animales y vegetales amenazadas o prioritarias como criterio de selección para la designación de zonas de especial conservación en Europa (European Commission 1992).

Gestión adaptativa y seguimiento de las actuaciones de conservación

La conservación de especies amenazadas requiere el análisis de la efectividad de las medidas aplicadas y de su repercusión en la dinámica poblacional (Margules y Pressey 2000, Groom et al. 2006). Una vez logrado dicho conocimiento, la probabilidad de mejorar su estado de conservación es mayor puesto que se podrá comprobar qué aspectos pueden mejorarse de entre los propuestos como medidas de actuación (Pullin et al. 2005, Arlettaz et al. 2010). Esta cuestión forma parte de la disciplina conocida como gestión adaptativa, que incorpora la investigación a la práctica de la conservación a través de la evaluación del diseño experimental de un proyecto, su gestión y su seguimiento, con el objetivo de adaptar las actuaciones a las mejores prácticas posibles y a los objetivos propuestos inicialmente, así como divulgarlo públicamente (Salafsky et al. 2001, Meffe et al. 2002).

En ocasiones, cuando el nivel de amenaza de una especie es muy elevado, es preciso llevar a cabo acciones de conservación antes de haber conseguido el conocimiento necesario. Ello supone reducir las garantías de consecución de los objetivos planteados (Pullin et al. 2004). En este caso, es preciso acompasar la investigación sobre biología, amenazas y patrones de selección de hábitat y alimentación, con la evaluación de la eficacia de las medidas de conservación aplicadas. Ello permitirá modular los planteamientos de manejo, inversión económica y líneas de actuación, en un marco de gestión adaptativa que resulta favorable y positivo (Salafsky et al. 2001, Meffe et al. 2002, Martín-López et al. 2009).

Las complejas relaciones hombre-conservación de especies

El hombre es el causante de la mayor parte de las amenazas que afectan a las especies silvestres, bien de forma intencionada o indirecta. Los factores que están haciendo reducir los niveles de diversidad biológica a escala global son, por orden aproximado, la destrucción y fragmentación de hábitats naturales, la introducción de especies exóticas invasoras, alteración de las características ecológicas de los territorios por efecto del cambio y calentamiento globales, la persecución directa, los cambios en los sistemas de aprovechamiento agrario hacia métodos intensivos con un mayor uso de productos fitosanitarios o la aparición de enfermedades emergentes (Wilson 1992, Brook et al. 2008). En tanto estos problemas no se eliminen o reduzcan, continuará acelerándose la tasa de extinción de especies en nuestro planeta (Hoffman et al. 2010).

Pero además de generar estos factores a gran escala, el hombre es un actor principal durante la ejecución de acciones de conservación y su papel debe tenerse en cuenta tanto para analizar en detalle las causas de declive como para la propuesta de acciones de conservación (Meffe et al. 2002). A menudo, la grave situación de determinadas especies surge como consecuencia de conflictos directos entre las especies y los intereses económicos o sanitarios de las personas (Luck et al. 2004, Heydon et al. 2010). Así, es preciso estudiar mecanismos de coexistencia sostenible para poder asegurar la resolución de los problemas. Esta es una línea de trabajo avanzada en numerosos países y que, en numerosas ocasiones, ha producido resultados satisfactorios (Young et al. 2005, Dickman 2010). En otras circunstancias, estos conflictos han conducido a acciones coercitivas para la población humana local o bien a la desaparición o declive de las especies que generaban el conflicto (Hazzah et al. 2009, New et al. 2011).

Dinámica, amenazas y conservación del gremio de buitres en España

Las especies de buitres que habitan la península Ibérica son el quebrantahuesos *Gypaetus barbatus*, el alimoche, el buitre negro y el buitre leonado. Estas cuatro especies conforman el gremio de rapaces necrófagas estrictas de España y presentan una ecología trófica y comportamental interrelacionada (Donázar 1993). Son especialistas en explotar un recurso como la carroña, rentable energéticamente por la abundancia en biomasa que ofrece y ausencia de conflictos violentos en su depredación, pero que conlleva elevados costes por la competencia intra e interespecífica en su aprovechamiento o por su impredecibilidad espacial y temporal (DeVault 2003, Ruxton y Houston 2004, Carrete et al. 2010, Barton et al. 2012). Los buitres ejercen tanto procesos facilitatorios para el hallazgo de carroñas (Jackson et al. 2008, Cortés-Avizanda 2010) como mecanismos de competencia por lugares de nidificación y alimento, o diferencias en los patrones de dispersión y migración (Cramp 1998). Las características morfológicas, de selección de hábitat y de movimientos migratorios de estas especies son consecuencia de adaptaciones evolutivas, habiendo determinado una complementariedad entre ellas en cuanto a la ocupación del territorio o a la explotación de las carroñas (König 1983, Hertel 1994, Demody et al. 2011). Así, la estructura del pico, el tamaño corporal, la presencia de plumas en la cabeza o la capacidad prensil de garras son indicadores de las preferencias por distintos tipos de restos (König 1983, Hertel 1994). Del mismo modo, diferencias en los patrones de campeo y prospección del territorio hacen que la dieta varíe en relación al tamaño de presa y a su carácter más o menos predecible respecto su distribución en el territorio (König 1983, Selva 2004). La territorialidad es otro componente que varía en los buitres ibéricos, mostrando un gradiente que va desde la ocupación y defensa de un territorio en el que satisfacer las demandas tróficas –casos del quebrantahuesos o alimoche- o abogar por criar en grupo y aumentar el intercambio de información para el hallazgo de carroñas– buitre leonado y buitre negro- (Donázar 1993, Demody et al. 2011). No obstante, esta relación no es lineal y tanto alimoches como buitres negros muestran patrones variables de territorialidad ó colonialidad. Del mismo modo, la agresividad y la tolerancia hacia otros congéneres es variable y hace que, por ejemplo, el quebrantahuesos disponga de territorios de dimensión variable en función de las fuentes de alimento existentes, lo que provoca en ocasiones el empaquetamiento de sus territorios (Carrete et al. 2006a, Margalida et al. 2008a, Cortés-Avizanda et al. 2010). En definitiva, las características de las cuatro especies de buitres ibéricos han propiciado una cohesión ecológica que les confiere funcionar en determinados casos como grupo/gremio dentro de la familia *Accipitridae*. Este gremio puede ampliarse con otras especies en el ámbito de la necrofagia, en el cuál se integrarían el resto de necrófagas facultativas (Selva y Fortuna 2007).

Servicios ecosistémicos prestados

Los ecosistemas naturales proporcionan elementos imprescindibles para que el hombre y el resto de especies puedan sobrevivir (Carpenter et al. 2009) Además de los elementos cuantificables derivados de la explotación de recursos naturales, los ecosistemas constituyen un reservorio para la sostenibilidad de la vida en el Planeta, tanto por lo que contienen en la actualidad como por la capacidad de regeneración de elementos materiales (Schröter et al. 2005). El estudio, evaluación y divulgación de los servicios que prestan los ecosistemas resultan necesarios para poner de manifiesto la importancia de conservar los procesos ecológicos (Dupont et al. 2011). Se trata de una disciplina de estudio compleja por la dificultad que entraña cuantificar dichos servicios, por la necesidad de establecer unidades de medida homogéneas y comparables, y por la propia complejidad de las relaciones ecológicas (Kremen 2005, Boyd y Banzhaf 2007).

Los buitres ibéricos prestan importantes servicios ecosistémicos, derivados principalmente de su papel como consumidores de biomasa de animales muertos y facilitadores de la eliminación de restos orgánicos del medio natural (DeVault et al. 2003, Şekercioğlu et al. 2004, Barton et al. 2012, Margalida y Colomer 2012). Los patrones temporales y espaciales de consumo de las carroñas dependen de numerosos factores, de modo que su evaluación constituye un reto para determinar las circunstancias en que los buitres pueden ser aliados del hombre para reducir la diseminación de enfermedades transmisibles, o para disminuir los costes económicos asociados a la recogida de cadáveres realizada (Caley y Hone 2004, Gortázar et al. 2007, Dupont et al. 2011, Ogada et al. 2012). Además, su integración en las cadenas tróficas como consumidor secundario contribuye a regular las relaciones con otros grupos taxonómicos mediante fenómenos de competencia –por ejemplo, con depredadores generalistas- o depredación –contribuyendo a capturar conejos moribundos- (Şekercioğlu 2006, Selva y Fortuna 2007). De esta manera, presenta un papel clave en el equilibrio de los balances energéticos de los ecosistemas ante alteraciones que puedan tener lugar, como son cambios en la abundancia de recursos tróficos, alteraciones de hábitat o epidemias sanitarias (Selva 2004, Markandya et al. 2008, Whelan et al. 2008). Por otro lado, además de su papel necrófago, los buitres generan beneficios en otros ámbitos. Desde el punto de vista social, constituyen un importante reclamo turístico en zonas rurales y atraen recursos a las economías locales (Wenny et al. 2011).

Tendencia y dinámica poblacional

España alberga las poblaciones de buitres más abundantes de todo el Paleártico occidental. A nivel europeo, buitres negros y leonados tienen más del 95% de su población reproductora en España, mientras que alimoche (80%) y quebrantahuesos (70%) también encuentran en España su principal área de distribución (Ministerio de Medio Ambiente y Medio Rural y Marino 2010). La tendencia poblacional de cada una de ellas ha sido diferente. Así, el buitre leonado y el buitre negro han experimentado importantes aumentos poblacionales en los últimos 30 años (Parra y Tellería 2004, Del Moral 2009a), llegando a alcanzar casi 25.000 parejas el primero y las 2.000 el segundo, desde poblaciones mucho más escasas en la década de los años 1970 (De la Puente et al.

2007, Moreno-Opo y Margalida *in rev*). El quebrantahuesos ha mostrado una tendencia positiva igualmente, aunque más atenuada y con unos efectivos poblacionales más escasos, existiendo en la actualidad 134 territorios regentados (Grupo de Trabajo del Quebrantahuesos 2012). El alimoche es la especie cuya tendencia ha sido menos positiva. Aunque los censos nacionales realizados han evidenciado un número de parejas aproximadamente estable, alrededor de las 1.400 parejas (Del Moral 2009b), el esfuerzo de prospección ha mejorado en los censos más recientes, hecho que ha podido introducir un sesgo sobre la tendencia poblacional. Al mismo tiempo, se han producido importantes declives en determinados núcleos poblacionales (Carrete et al. 2007, Margalida et al. 2012b). En un contexto global, la especie en un estado de conservación más desfavorable es el alimoche, por las amenazas que se ciernen tanto en las áreas de reproducción como de invernada (BirdLife International 2012). Por otro lado, España se localiza en el extremo occidental de distribución de estas especies y sus poblaciones, salvo para el alimoche, funcionan como metapoblaciones respecto a las continuas y abundantes de Oriente Medio y Asia Central (i.e. Agudo et al. 2011, Moreno-Opo et al. 2012b).

En relación a la dinámica poblacional es preciso tener en cuenta aspectos demográficos y ecológicos que determinan el estado de conservación y la viabilidad futura de la población (Moss et al. 1982). Así, en especies especialistas longevas y con bajas tasas de reproducción como son los buitres, es importante evaluar la influencia que tienen sobre la viabilidad de las poblaciones factores como la mortalidad no natural, la abundancia y dispersión de recursos tróficos, las tasas de productividad/éxito reproductivo, la disponibilidad de hábitat potencial, las características genéticas de la población o los patrones de dispersión y migratorios (Oro et al. 2008, Carrete et al. 2009a, García-Ripollés y López-López 2011).

Amenazas comunes al gremio de buitres: mortalidad y pérdida de hábitat

El aspecto más condicionante sobre la dinámica poblacional de los buitres es la mortalidad no natural de individuos adultos (Oro et al. 2008, Grande et al. 2009). De hecho, es el factor que ha provocado graves declives poblacionales en los últimos años en distintas regiones (Green et al. 2004, Koenig 2006, Virani et al. 2011). La causa que provoca mayor número de muertes no naturales es el envenenamiento por ingestión de cebos tóxicos, que se colocan mayoritaria e intencionadamente para matar especies depredadoras que entran en conflicto con intereses económicos de ganaderos y gestores cinegéticos (Carrete et al. 2007, Hernández y Margalida 2008, Mateo-Tomás et al. 2012), o bien como fármacos para combatir enfermedades del ganado (Green et al. 2004). En España, en general, los buitres no son las especies a las que se envenena intencionadamente –sí lo son zorros, lobos, perros o cualquier otro mesocarnívoro-, pero sufren las consecuencias debido a su capacidad de prospección del territorio y localización e ingestión de cebos tóxicos (Margalida 2012). Para todas las especies constituye la principal amenaza y para las que están en un estado de conservación más desfavorable es el factor que impide, en mayor medida, su recuperación poblacional (Madroño et al. 2004). En este sentido, los sectores sociales y organizaciones implicados en la conservación de los buitres son conocedores de la gravedad del asunto y numerosas iniciativas se han puesto en marcha para remediarlo (i.e. WWF España

2011, SEO/BirdLife 2012). No obstante, la heterogeneidad de casuísticas en el empleo de veneno, su carácter indiscriminado, la complejidad en la detección y disuasión de su uso, la dificultad para la identificación de responsables y la ausencia de alternativas viables, económicas y eficaces para el control de depredadores en el medio natural provocan que siga siendo un tema vigente y trascendente (Fundación Gypaetus 2005, Margalida 2012). Dentro de la intoxicación por compuestos químicos, el plumbismo representa otra amenaza que, si bien no provoca una mortalidad directa significativa, genera numerosos problemas de viabilidad reproductiva (Gangoso et al. 2009, Hernández y Margalida 2009a, Finkelstein et al. 2012).

Otros factores que provocan un número importante de muertes de especies de buitres en España son las colisiones con aerogeneradores y la electrocución-colisión con tendidos eléctricos (Camiña 2007, Tellería 2009a, 2009b, Carrete et al. 2012). La idoneidad de instalar parques eólicos es un asunto emergente en la conservación de la diversidad biológica, puesto que se enfrentan la posibilidad de emplear una energía no contaminante frente al impacto paisajístico y a la mortalidad causada en aves planeadoras (Noguera et al. 2010). En España constituye un riesgo muy importante para buitres leonados y alimoches en tanto se ubican en zonas clave de reproducción y migración (Carrete et al. 2012, Martínez-Abraín et al. 2012). De hecho, puede condicionar la viabilidad de determinadas poblaciones por su impacto continuado (Carrete et al. 2009a). La electrocución y colisión también afecta a los buitres aunque en menor medida que a otras rapaces cuyo comportamiento de campeo las hace más proclives a posarse en torretas eléctricas (Guil et al. 2011).

La alteración del hábitat representa otra amenaza para los buitres ibéricos, principalmente relacionada con los cambios en los usos agrarios que pueden determinar cambios en la disponibilidad de alimento (ver apartado siguiente). Los enclaves de nidificación, al situarse en roquedos y cantiles –salvo en el caso del buitre negro- evitan de manera general la pérdida de lugares potenciales de cría. No obstante, las molestias humanas causadas por actividades económicas o de ocio en las inmediaciones de los nidos pueden provocar el fracaso de la reproducción en función de la intensidad y frecuencia con que se produzcan (Arroyo y Razin 2006, Zuberogoitia et al. 2008).

Alteración en la disponibilidad de alimento

Un asunto al que han tenido que hacer frente las rapaces necrófagas en España en los últimos años ha sido la modificación de los patrones de aparición de carroñas en el campo, en relación a su ocurrencia y abundancia (Donázar et al. 2009a, Margalida et al. 2010). Los cambios normativos acaecidos tras la crisis de las vacas locas a primeros de los años 2000 obligaron a retirar de manera generalizada los cadáveres de ganado en cualquier tipo de explotación ganadera, permitiendo únicamente la posibilidad de destinar cadáveres a las aves necrófagas través de puntos de alimentación suplementaria vallados (García de Francisco y Moreno-Opo 2009). Esta situación fue generando situaciones negativas para las cuatro especies de buitres a medida que se implantó la recogida de cadáveres (Tella 2001, Donázar et al. 2009b). Algunas de las consecuencias de la reducción de carroña disponible y de la dependencia de fuentes predecibles de alimento son la alteración en los patrones de campeo y dispersión (Deygout et al. 2009, 2010, Margalida et al. 2010), cambios en la dinámica poblacional de las especies (Carrete et al. 2006a, Robb et al. 2008), el incremento del número de ejemplares ingresados en centros de recuperación por deshidratación y emaciación (García de Francisco y Moreno-Opo 2009) o cambios en la conducta alimentaria que suponen un riesgo para las aves –por alimentación continuada en basureros y por mayor proximidad a infraestructuras- o, incluso, alarma social por los ataques de buitres sobre ganado vivo (Donázar et al. 2010, Zuberogoitia et al. 2010, Margalida et al. 2011b). A la luz de esta situación, se ha modificado la legislación que regula la gestión de los subproductos animales no destinados a consumo humano, permitiendo en determinadas circunstancias que se puedan proporcionar cadáveres para la alimentación de especies necrófagas sin necesidad de recogerlos ni llevarlos a lugares fijos y vallados (Council of Europe 2009a, European Commission 2011). Se espera que estos cambios, una vez sean aplicados convenientemente, reestablezcan patrones de aparición de cadáveres de ganado equiparables a lo que ocurriría de forma natural y, en consecuencia, atenúen los efectos negativos causados (Margalida y Colomer 2012, Margalida et al. 2012a, Moreno-Opo et al. 2012c).

Además de lo anterior, otros asuntos relacionados con la disponibilidad de alimento se plantean de cara a futuro como condicionantes para la conservación de las especies de buitres. La pérdida de rendimiento económico de la ganadería extensiva está modificando la distribución y abundancia de las cabañas ganaderas, promoviendo las prácticas agrarias intensivas y disminuyendo las extensivas (Godfray et al. 2010). La población ganadera española ronda los 30 millones de cabezas de ganado, no habiendo sufrido cambios significativos durante los últimos 45 años (Spanish Statistics Institute www.ine.es). No obstante, la oveja ha reducido en nueve años su población un 25%, pasando de las 24.9 a los 18.5 millones (Spanish Statistics Institute www.ine.es). Este hecho es especialmente preocupante puesto que el ovino es la cabaña más abundante y la que más íntimamente está relacionada con la distribución y dieta de las especies de buitres, debido a su carácter extensivo y a su presencia en el entorno de las principales áreas naturales del país (De Juana y De Juana 1984, Margalida et al. 2009, Olea y Mateo-Tomás 2009). Por otro lado, la población de ungulados silvestres cinegéticos se ha incrementado en los últimos 20 años, a tenor de las estadísticas de caza disponibles (Spanish Statistics Institute www.ine.es , Garrido 2011), lo que ha supuesto un aumento importante de estas presas en la dieta de los buitres ibéricos (Corbacho et al. 2007). Esta cuestión podría considerarse positiva y paliativa de otros problemas si no fuera por el riesgo que entraña la alimentación de una carne que está expuesta a unos niveles muy elevados de contaminación por plomo (Taggart et al. 2011). Este hecho resulta condicionante y sus consecuencias sobre la dinámica poblacional no están muy claros, aunque los efectos negativos sobre la eficacia biológica sí han sido puestos de manifiesto, así como el riesgo de mortalidad asociado (Gangoso et al. 2009, Hernández y Margalida 2009a). Por último, el conejo es una especie clave en el funcionamiento de los ecosistemas ibéricos por ser una de las principales presas de toda la comunidad de depredadores (Delibes-Mateos et al. 2007). Sus cadáveres forman parte importante de la dieta de alimoches y buitres negros y, en menor medida, de quebrantahuesos y buitres leonados, durante los episodios anuales de mortandad por las epidemias víricas que padece en verano o en invierno-primavera (Costillo et al. 2007a, Donázar et al. 2010). Dichas enfermedades han reducido de manera importante su abundancia durante las últimas décadas (Gortázar et al. 2007, Delibes-Mateos et al. 2009) y han propiciado cambios en la composición de la dieta de los buitres (Costillo et al. 2007b).

Actuaciones realizadas y estado de conservación

Las cuatro especies de buitres se encuentran entre las que mayor atención han recibido por parte de investigadores, de modo que gran parte de los aspectos de su ecología y amenazas se han estudiado de manera muy detallada (i.e. Costillo 2005, Cortés-Avizanda 2010, Margalida 2010a). Además, han constituido buenos modelos de estudio para evaluar distintas hipótesis sobre comportamiento, evolución o estrategias de vida (Carrete et al. 2006b, Grande et al. 2008). Por ello, y posiblemente por su carácter llamativo y atractivo social, han sido receptores de numerosos programas de conservación. En este sentido, se conocen de manera adecuada los aspectos más generales e importantes que permiten desarrollar medidas de gestión efectivas (Moreno-Opo y Guil 2007, Donázar et al. 2009b). Así, estas especies han vehiculado la protección de espacios naturales por presencia de sus enclaves de reproducción, se han desarrollado proyectos de reintroducción para expandir su presencia (Simón et al. 2007, http://es.blackvulture-pyrenees.org/), se han aprobado planes de recuperación oficiales en distintas regiones españolas, se han promovido programas de alimentación suplementaria, se han aprobado planes de manejo ex situ y de cría en cautividad, han sido destinatarios de proyectos coordinados a nivel internacional y objeto de acciones contra el uso de venenos (SEO/BirdLife 2012). A pesar de todo ello, pocos trabajos han cuantificado y evaluado la eficacia de las acciones de conservación aplicadas sobre la dinámica poblacional o sobre el grado de amenaza (Oro et al. 2008, Margalida 2010b). En la actualidad, todas las especies están protegidas legalmente y no se pueden realizar acciones que dañen los individuos o sus hábitat (Ministerio de Medio Ambiente y Medio Rural y Marino 2011). El quebrantahuesos y el alimoche están considerados "en peligro de extinción", el buitre negro "vulnerable" y el buitre leonado "no amenazado" (Ministerio de Medio Ambiente y Medio Rural y Marino 2011), lo que significa que aún queda un largo camino por recorrer para mejorar su estado de conservación y reducir el nivel de amenazas que sufren.

El buitre negro: aproximación a su ecología

Características morfológicas e identificación

El buitre negro es la rapaz (orden Falconiformes, familia Accipitridae) más grande del Paleártico (Cramp 1998). Puede alcanzar los 10 kg de peso y una envergadura alar de 295 cm (Hiraldo 1977). Es una especie cuya coloración característica es marrón oscuro uniforme, presentando plumón negro o grisáceo en la cabeza, así como plumas lanceoladas en el cuello que forman la collareta lateral-posterior que le confiere el aspecto de monje que dio origen a su nombre en latín –*monachus*- (Moreno-Opo y Guil 2007). Es una especie monotípica, con escasas variaciones biométricas entre individuos de distintas áreas geográficas, siendo los presentes en la península Ibérica más pequeños (Cramp 1998, Del Hoyo et al. 1994).

Para su identificación en vuelo, resulta característico su gran tamaño y sus alas largas y anchas en forma rectangular de las que emergen, en el borde exterior, seis o siete plumas primarias. La cola es proporcionalmente corta y tiene forma de cuña ancha no muy apuntada. La cabeza no sobresale mucho de la silueta. Visualizado desde abajo, existe contraste en la coloración de las alas: las plumas cobertoras (la parte superior) son más oscuras que las primarias y secundarias (Moreno-Opo y Guil 2007, Del Moral y De la Puente 2010).

Las adaptaciones evolutivas han determinado la anatomía del buitre negro generando especializaciones en cuanto a su comportamiento trófico, organización social y selección de hábitat. Presenta un pico robusto y fuerte para desgarrar partes duras en los cadáveres (músculos, piel, tendones). Las plumas largas están ausentes de la cabeza, lo que evita que supongan interferencia por rotura, impregnación o reducción de la visibilidad durante la ingestión de partes internas en las carroñas (Donázar 1993). Tiene garras con cierta capacidad prensil, con dedos largos y uñas fuertes (Donázar 1993).

Datado y sexado

Las diferencias entre sexos son muy escasas, prácticamente imperceptibles en el campo, aunque las hembras son ligeramente más grandes que los machos (Newton 1979, Cramp 1998).

Los ejemplares juveniles presentan una coloración más oscura y mate en sus plumas, tanto en el cuerpo como en las alas, que torna progresivamente en tonos más pardos hasta alcanzar el patrón de adulto definitivo, alrededor de los seis años de edad (Hiraldo 1977, Forsman 2003). El color de la cabeza también varía con la edad (Del Moral y De la Puente 2010): los juveniles de primer año presentan plumón negruzco generalizado por toda la cabeza. En los inmaduros de segundo o tercer año el plumón negro de la cabeza se reduce coincidiendo con la apareción de franjas de piel desnuda que se desarrollan hacia las comisuras del pico y hacia la parte trasera de los ojos. En las aves de tres a cinco años, el plumón oscuro y tupido se va aclarando progresivamente y adquiere aspecto pardo claro. En estas fases intermedias (2º a 5º año) existe una gran variabilidad de coloración dentro de estos patrones generales (Moreno-Opo y Guil 2007). Los adultos poseen colores más aclarados, con cejas claras muy marcadas y apenas color negro-oscuro en la cabeza.

A través del estudio de la muda se pueden datar los ejemplares de hasta cinco años de edad, que es el tiempo que tardan en sustituir las últimas primarias juveniles (De la Puente y Elorriaga 2012). Muda las primarias de dentro hacia fuera, a razón de entre dos a cuatro plumas por año. Durante el primer año de vida no sustituye ninguna primaria y no es hasta la primavera-verano del segundo año cuando inician el recambio de primarias. Las aves de cinco años, que aún no han reemplazado las primarias juveniles más externas, vuelven a mudar las primarias internas (De la Puente y Elorriaga 2012).

Biología de la reproducción

El buitre negro presenta una estrategia reproductiva típica de especie longeva (Winemiller 1992), con una tasa de reproducción anual baja y una elevada supervivencia adulta, que puede llegar hasta los 35-40 años (Sánchez 2004). Alcanza la madurez sexual, entendido ésta como edad de primera reproducción, generalmente a los cuatro años (Tewes 1996), aunque existen referencias de aves criando en su tercer año calendario en el ámbito programas de reintroducción de la especie (Terrasse et al. 2004). Aunque se ha considerado una especie monógama (Cramp 1998) el radiomarcaje ha permitido evidenciar los cambios entre individuos de la pareja (Del Moral y De la Puente 2010) o incluso la formación de tríos de forma ocasional (Luque et al. 2010). Se reproducen cada año, dejando algunos de descanso intercalados en función de factores como la disponibilidad de alimento, la sustitución de alguno de los miembros de la pareja, la edad o la climatología (Hiraldo 1983). Los adultos permanecen ligados durante todo el año al nido, aunque durante fuera de la época de cría la presencia en torno al mismo resulta más esporádica (Bernis 1966).

El ciclo reproductivo dura aproximadamente nueve meses, y se inicia en enero con el cortejo. En esta época los nidos son muy frecuentados y se produce el arreglo y aporte de material, que se realiza con palos, lana, pelo y otro material vegetal. En ocasiones, la excesiva acumulación de material en los nidos provoca su desplome. Sitúa los nidos en la parte superior de la copa de los árboles, siendo éstos de un diámetro de entre 1,40-2,5 m y una altura de entre 0,9-1,3 m (Tewes 1996, Cramp 1998, Moreno-Opo y Guil 2007). Cada pareja regenta un número variable de nidos, con un promedio de 2,7 (De la Puente 2007). Las cópulas suelen producirse en los nidos a partir del mes de enero. Las puestas se inician a primeros de febrero y se prolongan hasta mediados de abril, siendo más numerosas en el período que transcurre entre la última semana de febrero hasta mediados de marzo (Del Moral y De la Puente 2010). Generalmente ponen un único huevo (Donázar 1993), que pesa entre 210,5 y 280,0 g (Glutz von Blotzheim et al. 1971). El período de incubación dura alrededor de 57 días, con un rango que va desde los 50 hasta los 68 días (Bernis 1966, Hiraldo 1983, Del Moral y De la Puente 2010).

El desarrollo post-embrionario es lento, dilatándose la estancia del pollo en el nido unos 110-120 días, (rango 88-137 días; Hiraldo 1983, Tewes 1996, Moreno-Opo y Guil 2007, Del Moral y De la Puente 2010). Durante las primeras semanas de vida del pollo, sobre todo durante las ocho primeras, los adultos permanecen junto a éstos en todo momento, reduciéndose su estancia paulatinamente a medida que los pollos crecen. Una actividad muy importante que los adultos realizan en nido es la de sombreo de los pollos, con objeto de protegerlos de la insolación y disminuir las posibilidades de deshidratación cuando éstos están aún cubiertos de plumón y carecen de plumas protectoras (Donázar 1993). Los primeros vuelos tras abandonar el nido transcurren desde inicios de agosto hasta primeros de octubre. En este período los juveniles regresan periódicamente al nido, vuelan junto a los progenitores y durante los dos primeros meses todavía reciben alimento de éstos (Hiraldo 1983).

Los parámetros reproductivos del buitre negro dependen en gran medida del impacto de factores que causan mortalidad no natural y de la disponibilidad de alimento (Sánchez 2004). La variable más empleada para valorar las tasas de reproducción es el éxito reproductivo (número de pollos volados/parejas que inician la reproducción). Los valores medios de éxito reproductivo en España se encuentran en torno al 0,75 pollos/pareja/año (Sánchez 2004), aunque existen diferencias entre distintas colonias y años. Las causas de esta diferencia son varias y dependen, por ejemplo, de la mortalidad no natural de las aves reproductoras (Sánchez 2004, Moreno-Opo et al. 2012a) o de cuestiones metodológicas: la fiabilidad de los parámetros está directamente ligada a la frecuencia de seguimiento (De la Puente 2006), por lo que si las visitas de censo a las colonias son escasas se tiende a sobreestimar los valores reproductivos. El éxito reproductivo también depende de las molestias de origen antrópico ocasionadas (Margalida et al. 2011c), de la disponibilidad de alimento y de la alteración sufrida por ésta en los últimos años (Costillo et al. 2007a, García de Francisco y Moreno-Opo 2009). La mayor proporción de fracasos se produce durante la incubación, sobre todo en los últimos días, cuando se evidencia la inviabilidad de los huevos (Del Moral y De la Puente 2010). Las muertes de los pollos son menos frecuentes y tienen que ver principalmente con la desaparición de algún progenitor, la ingesta de cebos envenenados o la deshidratación por falta de sombreo ante una molestia prolongada (Sánchez 2004, Margalida et al. 2011c).

Movimientos

El buitre negro es una especie sedentaria, cuyos ejemplares adultos permanecen la mayor parte del año, salvo durante el otoño, muy ligados a las colonias de cría (Bernis 1966). Los individuos juveniles, una vez independizados y durante los tres-cinco primeros años de vida, se desplazan distancias variables, visitando distintas áreas. Éstas pueden corresponder a otras colonias de cría, zonas con abundantes recursos tróficos o a desplazamientos junto a buitres leonados visitando zonas fuera del área de distribución en el norte de la península Ibérica o excepcionalmente llegando a zonas del África subsahariana (Del Moral y De la Puente 2010)

Los desplazamientos para la búsqueda de alimento son también variables. Sus áreas de alimentación dependen de la época del año, la edad y la actividad de los individuos (Newton 1979). La superficie promedio en la que se mueven los ejemplares para buscar alimento durante la época de cría varía de 61,000 ha a 151,000 ha, abarcando los movimientos diarios en torno a las 10.000 ha (Carrete y Donázar 2005, Costillo 2005, Vasilakis et al. 2006, Moreno-Opo et al. 2010). Los ejemplares no reproductores tienen áreas de alimentación mucho más extensas, acorde con los movimientos erráticos que realizan (Jiménez y González 2012). En esta época, tanto éstos como los adultos tienen también movimientos fijos dirigidos a lugares donde existe alimento predecible, como son los muladares (Moreno-Opo et al. 2010). Los movimientos de los buitres negros están condicionados por la climatología, como corresponde a aves planeadoras que dependen de la formación de corrientes térmicas para el planeo. El tiempo que dedican a la prospección es mayor si las condiciones son buenas, mientras que si existe lluvia constante o fuerte viento se ven limitados para localizar el alimento (Hiraldo y Donázar 1990).

Hábitat de reproducción y de alimentación

El buitre negro es una especie que en España selecciona hábitats forestales para nidificar, mientras que ocupa un espectro de ambientes más amplio para conseguir el alimento (Moreno-Opo y Guil 2007). Son varios los tipos de formaciones boscosas seleccionadas para criar: bosques mediterráneos maduros, con nidos situados en alcornoques *Quercus suber*, encinas *Quercus ilex*, enebros *Junniperus oxycedrus* y madroños *Arbutus unedo*, bosques supramediterráneos en *Pinus sylvestris* y *Pinus nigra*, bosques mediterráneos de coníferas *Pinus pinaster* y acantilados con vegetación en costas mediterráneas en *Pinus halepensis* (Morán et al. 2007, Del Moral y De la Puente 2010). Muestra preferencia por zonas alejadas de la presencia humana, prefiriendo árboles grandes en la parte superior de laderas con pendiente pronunciada y preferentemente no orientadas al norte (aunque depende de la disponibilidad de masas arbóreas) (i.e. Fargallo et al. 1998, Donázar et al. 2002, Morán-López et al. 2006). En amplias zonas de Asia, la especie construye los nidos en roquedos o en el suelo de laderas (Batbayar 2012, Sklyarenko y Katzner 2012).

Las áreas que el buitre negro prospecta para conseguir alimento son variadas y dependen en gran medida de la abundancia de presas y de la edad del ave (Costillo et al. 2007c). Así, los ambientes de matorrales abiertos, las dehesas y baldíos con matorral asociado son los que más frecuenta para detectar carroñas de lagomorfos, ganado doméstico o ungulados (Hiraldo 1977, Carrete y Donázar 2005, Costillo 2005). Existe variación de las áreas prospectadas en función de la época del año (Jiménez y González 2012). Así, durante la época de cría los buitres negros realizan desplazamientos alrededor de las colonias en los hábitat mencionados, en zonas con mayor cobertura de vegetación que las seleccionadas por el buitre leonado, por ejemplo (Costillo et al. 2007c). Durante el invierno los buitres negros se observan con mayor frecuencia en zonas de monte mediterráneo, en lugares donde se celebran cacerías que convierten el alimento en más predecible (Moreno-Opo y Guil 2007). Los ejemplares no reproductores realizan movimientos dispersivos donde el alimento resulta más abundante, ya sean sistemas agropecuarios ganaderos (dehesas y pastizales), bosques mediterráneos maduros con abundancia de ungulados silvestres o conejo, o muladares (Moreno-Opo et al. 2010, Jiménez y González 2010)

Dieta

La mayor parte de la dieta del buitre negro la componen mamíferos (Corbacho et al. 2007, Costillo et al. 2007a), existiendo variación en el tipo de carroña explotada, según la disponibilidad, el área geográfica y la época del año. Prefiere piezas no muy grandes (Hiraldo 1977), para evitar la interacción con los buitres leonados que sí seleccionan cadáveres de grandes herbívoros (Donázar 1993, Hertel 1994). En las carroñas selecciona las partes más duras, como tejidos musculares y cartilaginosos (König 1983, Donázar 1993). El conejo tiene gran importancia en la dieta del buitre negro en España, con porcentajes de aparición en su dieta desde el 3% hasta el 60% (véase revisión en

Corbacho et al. 2007). La variación en las abundancias de conejo, debido en parte a su disminución poblacional por enfermedades (Delibes-Mateos et al. 2007, Gortázar et al. 2007), ha propiciado un cambio en la dieta en los últimos años, reduciendo en amplias zonas su consumo (Costillo et al. 2007a). Los restos de ungulados silvestres cinegéticos, muchos procedentes de los despojos de cacerías invernales, son también importantes y han adquirido un papel cada vez más representativo en la dieta del buitre negro (Corbacho et al. 2007). Los cadáveres de ganado doméstico, sobre todo de oveja, son un recurso trófico muy importante para el buitre negro (Corbacho et al. 2007, Costillo et al. 2007a) por lo que resulta una cabaña a proteger en actuaciones de gestión (Garzón 1997, García de Francisco y Moreno-Opo 2009). Los muladares tienen también importancia como áreas fijas de obtención de alimento, resultando fundamentales para los individuos reproductores de algunas colonias de cría (Del Moral y De la Puente 2010), así como para un importante contingente de ejemplares inmaduros (Cortés-Avizanda et al. 2010). En relación a los cambios en la disponibilidad de alimento producidos en los últimos años, sobre todo por la reducción de conejos y la alteración en la aparición de cadáveres de ganado (Donázar et al. 2009a), los buitres negros han mostrado una capacidad de adaptación a dichos cambios y una consiguiente plasticidad en su dieta (Costillo et al. 2007b).

Se han estimado unos requerimientos nutricionales de 500-656 g de carroña/día (Donázar 1993), que puede obtener de forma irregular gracias a la capacidad de ingerir de una vez en torno a 1,5 kg de biomasa. El período en que necesita mayor consumo de alimento transcurre entre la mitad y el final del crecimiento de los pollos, con unos 700 g diarios (Hiraldo 1983). Por ello, se estima que una pareja de buitres negros reproductores necesita unos 600 kg de carne cada año (Donázar 1993).

Comportamiento y organización social

El buitre negro se reproduce en colonias laxas. Esto es, los nidos se concentran en núcleos delimitados aunque distantes entre sí. Por otro lado, la estructura interna de la colonia funciona como un entramado de territorios que los buitres negros defienden variablemente (Bernis 1966, Moreno-Opo y Guil 2007). A una distancia que puede variar desde decenas hasta varios cientos de metros se encuentran los nidos de las parejas más cercanas. Los buitres negros aprovechan los beneficios de ambos tipos de uso del espacio; por un lado, la cercanía entre parejas permite la comunicación entre individuos sobre hallazgos de grandes presas mientras que conserva el comportamiento territorial para la prospección de presas de menor tamaño, que se distribuyen más uniformemente en el territorio (Donázar 1993). La distancia entre colonias depende del hábitat disponible y su superficie puede extenderse por varios cientos de kilómetros cuadrados o por escasas decenas. Incluso, se ha documentado la existencia de parejas solitarias (Del Moral y De la Puente 2010).

Los ejemplares adultos permanecen asociados durante la práctica totalidad del año a las colonias. Los ejemplares inmaduros, en cambio, pueden permanecer ligados en torno a una fuente de alimento por un tiempo variable. Así, los buitres negros pueden utilizar dormideros comunales, cuyo sustrato principal son árboles y roquedos (Cramp 1998). Suelen ser compartidos con buitres leonados (Bernis 1966, Valverde 1966) y esporádicamente con inmaduros de águilas imperiales *Aquila adalberti*, reales *Aquila chrysaetos* o perdiceras *Aquila fasciata*. De estas congregaciones obtienen beneficios, puesto que reciben información sobre áreas donde existe alimento, ahorrando energía y estableciendo vínculos sociales (Donázar 1993).

El buitre negro tiene, en general, un comportamiento poco agresivo. Los territorios de nidificación son defendidos por ambos miembros de la pareja, expulsando a otros congéneres en la época de celo e incubación. Las plataformas de nidificación son usurpadas en ocasiones por buitres leonados, existiendo cierta competencia en zonas donde este último resulta muy abundante o carece de sustrato de nidificación suficiente. A escala local este hecho puede llegar a ser importante y afectar negativamente al buitre negro (Traverso 2001).

Distribución y tendencia poblacional

A nivel mundial, la especie se distribuye por zonas templadas del Paleártico, desde la península Ibérica hasta China, Rusia y península de Corea (Del Hoyo et al. 1994), de manera discontinua. La población mundial ronda las 7.200-10.000 parejas reproductoras (BirdLife International 2012). En Europa crían unas 2.100 parejas (Moreno-Opo y Margalida *en rev.*) ubicadas en España, Portugal, Grecia y Francia. La tendencia general de las poblaciones en Europa es de incremento. En Asia existe un desconocimiento general sobre el número de parejas existentes, aunque distintas estimas realizadas indican 5.500-8.000 y muestran la importancia, por un lado, del extenso núcleo poblacional de Mongolia, que es el país con mayor número de parejas (4.000, Batbayar 2012), China y países del centro de Asia (Kazajstan, Uzbekistan, Tajikistan, Kyrgizistan, Turkmenistan) y, por otro, del entorno de países caucásicos (Armenia, Georgia y Azerbaijan principalmente) y Turquía (Dobado y Arenas 2012). La tendencia global de la especie es negativa y por ello está considerada *Casi Amenazada* (BirdLife International 2012).

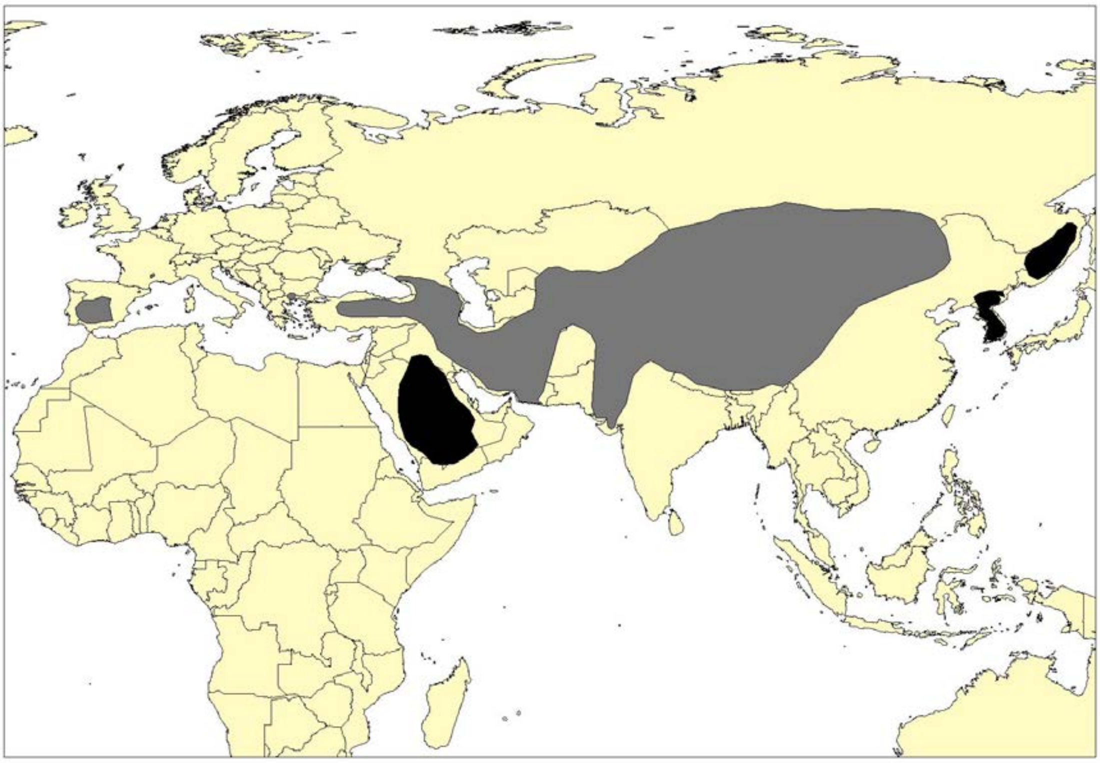

Figura 1. Área de distribución aproximada de las poblaciones reproductoras de buitre negro *Aegypius monachus* en 2012. En gris se exponen poblaciones reproductoras e invernantes, y en color negro las áreas de invernada (Fuente: BirdLife International 2012 y datos propios).

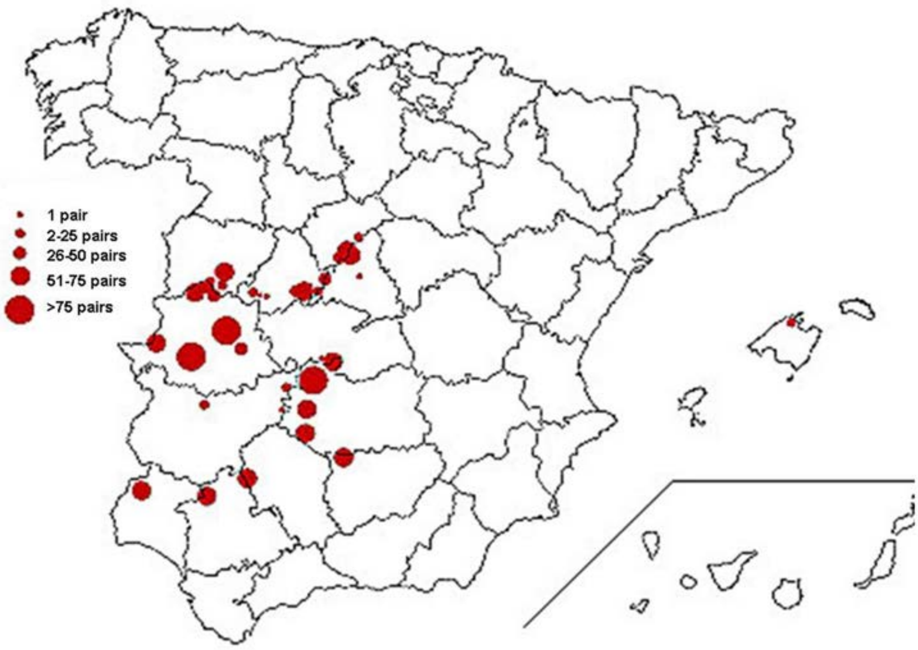

Figura 2. Núcleos y colonias de cría de buitre negro *Aegypius monachus* en España peninsular excepto Cataluña, mostrando el número de parejas reproductoras. (Fuente: De la Puente et al. 2007)

Tabla 1. Parejas de buitre negro *Aegypius monachus* en los distintos países donde se ha comprobado su nidificación en los últimos diez años (Fuente: Skartsi et a. 2008, LPO 2010, BirdLife International 2012, Dobado y Arenas 2012, Moreno-Opo y Margalida *en rev.*) * = individuos.

País	*Parejas reproductoras (n)*	*Año*	*Tendencia*
España	2.068	2011	positiva
Francia	23	2010	positiva
Portugal	3	2011	positiva
Grecia	19	2005	estable
Ucrania	11	2004	negativa
Rusia	> 1.000	2002	negativa
Turquía	200-300	2004	negativa
Armenia	7-8	2002	negativa
Azerbaijan	10-30	2002	negativa
Georgia	20-30	2001	negativa
Kazajstán	150-300	2004	desconocida
China	> 1.760 *	1991	desconocida
Mongolia	4.000	2004	desconocida
Kazajstán	400-500	2004	desconocida
Uzbekistán			
Turkmenistán			
Tadyikistán			
Kirguizistán			
Irán	desconocido		desconocida
Afganistán	desconocido		desconocida
India	desconocido		desconocida
Pakistán	desconocido		desconocida
Total estimado	7.200-10.000		negativa

En España, el buitre negro cría asociado a formaciones montañosas de siete regiones: Extremadura, Castilla-La Mancha, Castilla y León, Andalucía, Madrid, Islas Baleares y Cataluña. En esta última región, las parejas asentadas (tres en 2012) provienen de un proyecto de reintroducción (www.blackvulture-pyrenees.org). Los censos realizados en España han mostrado un paulatino incremento de las poblaciones de la especie. El primer censo realizado dio como resultado el hallazgo de 206 parejas en 1973, aunque la cobertura del censo se considera incompleta (Hiraldo 1974). Posteriormente, se produce un incremento paulatino hasta el censo nacional coordinado de 2006, en el que se hallaron 1625 parejas reproductoras (De la Puente et al. 2007). La última recopilación de censos, para 2011, otorga una cifra de 2068 parejas reproductoras (Moreno-Opo y Margalida *en rev.*) La mayoría de las colonias han tenido un crecimiento poblacional destacable, sobre todo en Extremadura, Madrid y Castilla y León.

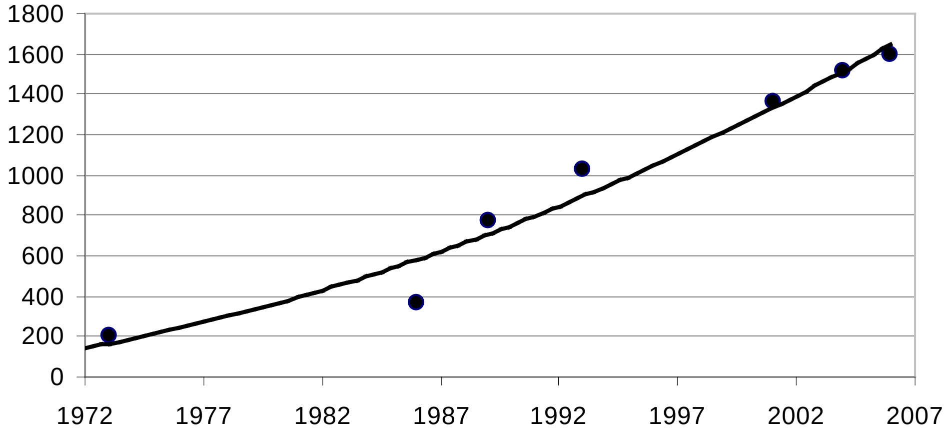

Figura 3. Evolución de la población reproductora de buitre negro *Aegypius monachus* en España (número de parejas reproductoras). (Fuente: Moreno-Opo y Guil 2007)

OBJETIVOS

Esta Tesis Doctoral está dirigida al estudio de las relaciones del buitre negro con su hábitat de reproducción y alimentación, así como a evaluar los efectos de determinadas actividades humanas. Los resultados son discutidos desde una perspectiva práctica, sugiriendo la aplicación de medidas de actuación que mejoren su estado de conservación. En este sentido, se toman como modelo de estudio tanto los individuos como las poblaciones de buitre negro, teniendo en consideración, en la medida de lo posible, su dinámica poblacional y los procesos ecológicos que ocurren en el área de estudio. Los objetivos de la Tesis Doctoral son:

- Profundizar en la comprensión de las relaciones entre el buitre negro y el hábitat en el que se localiza, con el propósito de identificar las características del medio más importantes para la reproducción –tomando como referencia la localización de los nidos y el éxito reproductivo- y para la alimentación –de acuerdo con patrones de campeo y de selección de distintos tipos de presa-. Este objetivo se desarrolla en los capítulos 1 a 4, en la sección *Reproducción, dieta y selección de hábitat en el buitre negro*.
- Basado en lo anterior, evaluar las consecuencias de determinadas actividades antrópicas sobre las áreas de reproducción y alimentación del buitre negro, con objeto de establecer patrones de gestión y aprovechamiento compatibles que eviten los efectos negativos sobre la especie objetivo. Esta cuestión se expone en los capítulos 5 y 6, en la sección *Gestión del hábitat de reproducción y alimentación*.
- Analizar y actualizar el estado de conservación del buitre negro en España, así como los factores que han podido determinar su dinámica poblacional en los últimos años, a través de la recopilación de bibliografía sobre el esfuerzo de investigación realizado y sobre medidas de conservación aplicadas. Este punto se desarrolla en el capítulo 7, relativo al *Estado de Conservación*.

Reproducción, dieta y selección de hábitat en el buitre negro (capítulos 1 a 4)

El buitre negro es una especie para la cuál existe un amplio conocimiento sobre patrones de selección del hábitat de nidificación (i.e. Fargallo et al. 1998, Poizaridis et al. 2004, Donázar et al. 2002, Morán-López et al. 2005, Gavashelishvili et al., 2006). Esta información ha sido emitida para distintas localidades de su área de distribución Paleártica, principalmente en España. Sin embargo, ningún estudio ha abordado un análisis comparativo a escala global de los resultados de selección de hábitat de reproducción, considerando asimismo la influencia que tiene el empleo de distintas metodologías de trabajo sobre los resultados finales. Este asunto es tratado en el capítulo 1 de la presente Tesis Doctoral. En primer lugar, se llevó a cabo un trabajo de campo para identificar los factores que determinaron la selección de hábitat de nidificación en la colonia de cría de la Umbría de Alcudia, Castilla-La Mancha (España Central). Con posterioridad, los resultados de este estudio de campo junto con los de otras siete publicaciones que informaron sobre 15 colonias de cría fueron analizados para reconocer los factores que determinan la selección de hábitat de reproducción a escala global. Del mismo modo, se analizó la influencia que tiene el tipo de procedimiento experimental y de toma de datos sobre la variabilidad explicada por los modelos resultantes (expresada como varianza o devianza). De esta manera, es posible proponer metodologías más adecuadas en programas de investigación coordinados para la misma especie a escala global.

En el capítulo 2 se discuten los factores que propiciaron un mayor éxito reproductivo del buitre negro. Se llevó a cabo un muestreo de campo de las características geomorfológicas, ecológicas y de vegetación, así como de variables sobre la influencia del hombre, a escala de microhábitat y de paisaje, con el propósito de modelar los factores más influyentes sobre la mayor probabilidad de éxito en la reproducción, tomando como referencia la colonia de la Umbría de Alcudia, Castilla-La Mancha (España Central). Los resultados se compararon con los trabajos previos que abordaron esta cuestión para el buitre negro en España (Donázar et al., 2002, Morán-López et al. 2006).

El conocimiento del uso del espacio que ejercen las especies constituye un aspecto primordial de cara al planteamiento de medidas de conservación. En este sentido, y tomando como referencia los estudios previos sobre selección de hábitat de alimentación, se planteó la búsqueda de una nueva metodología de evaluación de áreas de campeo que, de manera ágil y con bajo coste, permitiera realizar una aproximación fidedigna a las zonas que ofrecen recursos tróficos para la especie. Así, en la misma colonia de cría se recogieron sistemáticamente crotales de ganado en las inmediaciones de nidos y posaderos de buitre negro, al objeto de conocer su origen geográfico y localizar de forma precisa –a escala de explotación ganadera- dónde los buitres los ingirieron junto con el cadáver. El estudio de las áreas de campeo se realiza normalmente individualizando el ejemplar objeto de estudio, mediante radiotelemetría o con transmisores satélite, GPS o GSM, de modo que unos pocos individuos informan sobre las áreas de campeo y alimentación (Carrete y Donázar 2005, Jiménez y González 2012). A diferencia de lo anterior, en este trabajo, incluido en el capítulo 3, se usó una novedosa metodología, más limitada en cuanto a la variedad de tipos de presa estudiadas –solo incluyó ganado- pero más genérica en relación al número de aves para el que se ofrecen resultados –las pertenecientes a dicha colonia- y más precisa sobre la localidad donde se alimentaron los buitres negros.

Distintos estudios previos han analizado la dieta del buitre negro basándose en restos de presas hallados en egagrópilas (Corbacho et al. 2007). Así, existe un adecuado conocimiento de los patrones de variación temporal y geográfica de la dieta, así como de la importancia de distintas especies en ella (Corbacho et al., 2007, Costillo et al. 2007a, 2007b). Sin embargo, hasta la fecha ningún estudio ha evaluado de manera detallada los factores que influyen sobre la presencia del buitre negro en carroñas, en las que se haya controlado el tipo de aportes suministrados mediante un diseño experimental adecuado. Este es el tema tratado en el capítulo 4 de esta Tesis Doctoral, en el que se identificaron las características de los aportes de carroña para evaluar diferencias en las tasas de alimentación de buitres negros. Así, la obtención de información sobre este particular es clave para el manejo y optimización de los puntos de alimentación suplementaria, de modo que puedes dirigir la alimentación a la especie que interesa ayudar desde el punto de vista de su conservación. Este trabajo se realizó en distintas zonas de Extremadura y Castilla-La Mancha (España occidental y central, respectivamente), en el entorno de núcleos de nidificación de buitre negro.

Gestión del hábitat de reproducción y alimentación (capítulos 5 y 6)

El efecto que tienen distintas actividades socioeconómicas sobre especies amenazadas ha sido objeto de estudio tanto en España (Bautista et al. 2004, González et al. 2006a, Zuberogoitia et al. 2008) como en otros países (i. e. Richardson y Miller 1997, Young et al. 2005, Preisler et al. 2006). En relación a la reproducción en aves, el estudio de las molestias humanas ha permitido comprender los efectos negativos que pueden llegar a tener sobre distintas variables poblacionales (Blumstein et al. 2005, Gill 2005) aunque para rapaces amenazadas la información disponibles es aún escasa (Richardson y Miller 1997, Arroyo y Razin 2006, González et al. 2006a, Zuberogoitia et al. 2008). Del mismo modo, ante la presencia humana, se ha propuesto de manera general el establecimiento de zonas de protección (buffer zones) alrededor de lugares clave en las que se prohíban o limiten ciertas actividades humanas durante determinados periodos (Fernández-Juricic et al. 2005, González et al. 2006a). No obstante, existen algunos aprovechamientos que son fundamentales para el mantenimiento de las economías locales y la preservación del medio natural. Es el caso del descorche en las áreas de cría del buitre negro, asunto tratado en el capítulo 5. En la colonia de cría de la Umbría de Alcudia, Castilla-La Mancha (España Central), se analizó el impacto de dicha actividad sobre el éxito reproductivo del buitre negro y sobre las reacciones generadas en los propios individuos.

Los cambios legales relativos a la gestión de subproductos animales con fines de alimentación de aves necrófagas que se implantaron a primeros de los años 2000 provocaron la alteración en la disponibilidad y aparición de cadáveres en el campo (Tella 2001, Donázar et al. 2009a, Margalida et al. 2010). Al mismo tiempo, estos cambios provocaron la aparición de efectos negativos sobre la dinámica poblacional y el comportamiento de determinados individuos o poblaciones (Donázar et al. 2009b, Margalida et al. 2011b). Por ello, la legislación relativa al control y gestión de subproductos animales ha sido modificada con el propósito de armonizar la seguridad sanitaria de animales y personas y la alimentación de especies necrófagas de manera sostenible y natural (European Commission 2011, Margalida et al. 2012a). En este sentido, podría resultar necesario, en determinadas circunstancias, usar técnicas y modelos de vallados de puntos de alimentación que faciliten el manejo de la alimentación de rapaces necrófagas a propietarios y gestores de fincas privadas y explotaciones ganaderas, administraciones u otras organizaciones y personas interesadas. Con este fin, se evaluaron distintos métodos-modelos de delimitación de puntos de aporte de carroña en relación a su eficacia para, por un lado, permitir la entrada de rapaces necrófagas y, por otro, evitar la ingestión de carroña por especies no objetivo potenciales vectores de enfermedades. Además de lo anterior, se consideró un elemento favorable la facilidad de transporte e instalación del modelo, su carácter asequible y su aplicabilidad en distintas regiones.

Estado de conservación (capítulo 7)

En conservación de especies, es importante evaluar si las medidas de conservación aplicadas han resultado efectivas y han tenido el impacto positivo inicialmente planteado (Groom et al. 2006). Este análisis revela si el enfoque práctico de actuación habría de modificarse y si existen lagunas de conocimiento que deberían ser cubiertas (McCarthy y Possingham 2007, Arlettaz et al. 2010). La recopilación y análisis de bibliografía puede contribuir a analizar el esfuerzo de seguimiento e investigación dedicado a una especie (Zhang et al., 2010, Liu et al. 2011). Además, este esfuerzo de investigación podría correlacionarse con el estado de conservación de la especie, su tendencia poblacional y las amenazas que la afectan. Hasta la fecha, pocos estudios han realizado este ejercicio para especies amenazas. Para el buitre negro en España, este análisis, junto con la actualización de su estado de conservación, se presenta en el capítulo 7.

CAPÍTULOS

CHAPTERS

REPRODUCCIÓN, DIETA Y SELECCIÓN DE HÁBITAT EN EL BUITRE NEGRO

BREEDING, DIET AND HABITAT SELECTION OF THE CINEREOUS VULTURE

CAPÍTULO-CHAPTER
1

Effect of methodological and ecological approaches on heterogeneity of nest-site selection of a long-lived vulture

by

Rubén Moreno-Opo, Mariana Fernández-Olalla, Antoni Margalida, Ángel Arredondo & Francisco Guil (2012)

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Efecto de la metodología de estudio y de la influencia de factores ecológicos sobre la heterogeneidad en la selección de hábitat de nidificación de una especie longeva de buitre

La aplicación de medidas de conservación basadas en evidencias científicas requiere que las metodologías de muestreo en estudios de modelización sobre aspectos ecológicos similares produzcan resultados comparables que faciliten su interpretación. El presente estudio plantea mostrar cómo la elección de distintos enfoques metodológicos y ecológicos puede condicionar las conclusiones de los estudios de selección de lugares de nidificación a lo largo de distintas metapoblaciones paleárticas de una especie indicadora. En primer lugar, se realizó un análisis multivariante acerca de las variables que determinaron la selección de los lugares de nidificación en una colonia de cría de buitre negro en España Central. A continuación, se aplicó un metanálisis para evaluar cómo los factores metodológicos y las características del hábitat determinan las diferencias y similitudes en los resultados obtenidos en estudios previos que modelizaron el hábitat de reproducción forestal de la especie. Los resultados revelaron patrones comunes de selección de hábitat de nidificación del buitre negro en toda el área de distribución estudiada, dirigidos positivamente hacia laderas de pendientes pronunciadas y orientadas preferentemente hacia el sur, elevada cobertura de grandes árboles y largas distancias a puntos con actividad humana. La situación de los puntos de muestreo aleatorio respecto a la colonia de cría, la proporción entre nidos y puntos aleatorios, el uso de puntos respecto polígonos como unidades de muestreo y el número de años de toma de datos determinaron la variabilidad explicada por el modelo. Además, en colonias de cría de gran tamaño las variables ecológicas y geomorfológicas resultaron más influyentes sobre la selección de hábitat de nificación que en colonias medianas y pequeñas. Del mismo modo, las actividades humanas afectaron en mayor proporción a las colonias situadas en bosques de tipo mediterráneo. Por primera vez, se ha aplicado un metanálisis relativo a los factores que influyen en la heterogeneidad de la selección de emplazamiento de los nidos de una especie particular, a una escala geográfica amplia. En este sentido, es esencial homogeneizar y coordinar el diseño experimental en la modelización de los requerimientos ecológicos de las especies, al objeto de evitar que las diferencias en los resultados entre estudios se deban a variaciones en los procedimientos metodológicos. Ello optimizaría la aplicación de las mejores prácticas de conservación y gestión de especies en un contexto global.

Effect of Methodological and Ecological Approaches on Heterogeneity of Nest-Site Selection of a Long-Lived Vulture

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Abstract

The application of scientific-based conservation measures requires that sampling methodologies in studies modelling similar ecological aspects produce comparable results making easier their interpretation. We aimed to show how the choice of different methodological and ecological approaches can affect conclusions in nest-site selection studies along different Palearctic meta-populations of an indicator species. First, a multivariate analysis of the variables affecting nest-site selection in a breeding colony of cinereous vulture (*Aegypius monachus*) in central Spain was performed. Then, a meta-analysis was applied to establish how methodological and habitat-type factors determine differences and similarities in the results obtained by previous studies that have modelled the forest breeding habitat of the species. Our results revealed patterns in nesting-habitat modelling by the cinereous vulture throughout its whole range: steep and south-facing slopes, great cover of large trees and distance to human activities were generally selected. The ratio and situation of the studied plots (nests/random), the use of plots vs. polygons as sampling units and the number of years of data set determined the variability explained by the model. Moreover, a greater size of the breeding colony implied that ecological and geomorphological variables at landscape level were more influential. Additionally, human activities affected in greater proportion to colonies situated in Mediterranean forests. For the first time, a meta-analysis regarding the factors determining nest-site selection heterogeneity for a single species at broad scale was achieved. It is essential to homogenize and coordinate experimental design in modelling the selection of species' ecological requirements in order to avoid that differences in results among studies would be due to methodological heterogeneity. This would optimize best conservation and management practices for habitats and species in a global context.

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Introduction

To date numerous studies have evaluated the relationships between one or various threatened species and the environmental variables at work in the habitats in which they carry out the distinct phases of their life cycles [1]. Of these, one of the commonest areas of study is research into the factors affecting reproductive processes, which have serious repercussions for population dynamics, and which have become one of the most important lines of work in conservation biology [2,3]. In general, the extent of our knowledge of the factors that determine reproduction depends on natural processes and/or human activities, but it is also influenced by the methodology employed in research [4,5]. Thus, general conclusions regarding the ecological aspects affecting the choice of reproduction sites of a single species at different spatial scales and geographical locations has only been possible in a very few cases. A possible solution is the application of a meta-analysis in order to combine results of previous studies and draw general conclusions concerning the ecological and human factors that affect habitats and species under study [6]. In order to do so, this type of analysis has to overcome the difficulties posed by the need to standardize heterogeneous information, the deficiencies in data collection in certain analyses, the lack of unifying criteria in data recording and variations in the ecological requirements of the species being modelled [7,8].

In light of these considerations and taking as a case study the cinereous vulture (*Aegypius monachus*) we evaluated the factors that determine breeding habitat selection by a new field study in Spain. This species is a good model for evaluating the conservation status of the ecosystems in which it breeds given its role in trophic chains by completing the processing cycle and assimilation of biomass of dead animals [9,10] and its sensitivity to alterations affecting the landscapes it inhabits, such as non-compatible forestry practices or human disturbances [11,12].

From a descriptive perspective, precedent studies showed common patterns of nesting-habitat selection by cinereous vulture [13–19]: nests were located in forests situated on mountain slopes with large trees and high vegetation cover, far away from the human presence. Nevertheless, there are also differences across the studies in the variables that were statistically significant as well as divergences in the applied methods that could affect the final results. Therefore, we evaluated as hypothesis how the methodological procedures influenced the variability reflected in the results of different studies and, as consequence, the nesting-habitat selection of the cinereous vulture at a global scale, through meta-analysis [20]. Meta-analysis is a statistical procedure for combining data from multiple studies by applying objective formulas with the purpose of evaluating the identification and reasons of the common findings or the variation among the results of the compared studies [20].

The general objectives of this work were thus:

i) to know the environmental factors determining the nesting-habitat selection of the cinereous vulture in a breeding colony of central Spain,
ii) to study the causes of variation in results regarding nest-site preferences from different published studies, together with the present field study, all integrated in a meta-analysis, and
iii) to evaluate which of the statistically significant factors highlighted in each study are the most relevant to nest-site selection in a Palearctic context and how they are related to the vulnerability and the ecological characteristics of each studied population, as a way of establishing the most appropriate management and conservation measures.

Methods

The study species

The cinereous vulture is classified as near threatened (7 200–10 000 pairs) [21] and breeds from the Iberian Peninsula as far as Eastern Asia. This vulture can be considered as an habitat indicator species due to its large foraging range [9], the specificity of its food requirements [22,23] and its nest-site selection in large mature trees [13]. This species' habitat is located in areas with high conservation status that are important to many other species, some of which are also threatened [24–26].

Study area

The nest-site selection study was conducted in Alcudia and Sierra Madrona Natural Park, Spain (Figure 1), home to a colony of 129 pairs [27]. The site is part of an upland area (736–1 115 m a.s.l.) in which the dominant vegetation consists of typical Mediterranean trees such holm oak (*Quercus rotundifolia*), cork oak (*Quercus suber*), strawberry-tree (*Arbutus unedo*), prickly juniper (*Juniperus oxycedrus*) and Lusitanian oak (*Quercus faginea*), associated with a well-developed shrub layer.

Field work and studied variables

In October-December 2005, once the breeding season was over, we visited all the cinereous vulture nests in the area (n = 155 nests). All nests, occupied in 2005 or unoccupied but with evidence of occupation in recent years, were studied [14,28]. We also selected random points (n = 85) in areas within the perimeter of the breeding colony [15,29]. We visited each point and recorded, from the nearest tree (since cinereous vulture breed in trees), the same data as for nests. The presence or absence of a nest at each point was established as a response variable [30].

The independent variables studied were chosen on the basis of previously evaluated aspects of this species' nest-site selection [15,16] or as factors that relate to the land management [15]. Specifically, explanatory variables affecting factors relating to two spatial scales (microhabitat, n = 3, and landscape, n = 18) were selected.

As microhabitat variables we measured the tree characteristics in relation to the tree species (*Sp_tree*), the height (m) from a visual estimation of the tree where nest is present or the tree randomly selected (*H_tree*) and the diameter (cm) at breast height (dbh) of tree where nest is present or the tree randomly selected (*D_tree*).

For the landscape scale we considered 1) geomorpholocial variables as the altitude (m asl, *Alt*), the presence of nest/random tree in a natural scree (yes/no, *Scree*), the orientation of the slope where nest/random tree is located since all nests in the study area are situated in slopes greater than 15% (N, S, W, E, *Orient*), the slope of the hillside in a 100 m radius around the location of a nest/random tree (%, *Slope*) and the distance (m) from nest/random tree to nearest natural scree (*D_scree*), all of them calculated through GIS (ArcView 3.1 software) and aerial photographs; 2) vegetation variables, as the number of trees taller than 4 m existing in a 25 m radius around the nest/random tree through setting a survey plot and visual estimation of the height of the trees (*Rad25_tree*), the average high of the shrub in a 100 m radius around the nest/random tree through the measurement of the shrubs existing in four line-transects (*H_shrub*), the percentage of coverage in a 100 m radius around nest/random tree of trees (*%_tree*), shrub (*%_shrub*), pasturelands (*%_past*), scree or rock outcrop (*%_scree-rock*), cork oak tree (*%_Qsuber*), holm oak tree (*%_Qrot*) and other tree species (*%_othersp*) through a direct visual assessment in the field; and 3) human disturbance-related variables as the length (m) of unpaved tracks in a 500 m radius around the nest/random tree (*Long_tracks*) and the distances (m) from the nest/random tree to the nearest paved road (*D_road*), to the nearest building (*D_const*) and to the nearest unpaved track (*D_track*) through the application of geographic information systems (GIS, ArcView 3.1) to measure distances and length of tracks.

Climatic factors were not taken into account due to the relatively small study area (11 115 ha) and the homogeneity of vegetation structure and altitude intervals, which imply similar values of temperature, rainfall, humidity or solar radiation across the studied landscape.

Data compilation from previous studies: meta-analysis

We carried out a bibliographical search of articles published on habitat selection in the cinereous vulture in peer-reviewed journals and official reports. We were able to collate data from seven articles referring to 15 colonies in three different metapopulations (Figure 1, see Appendix S1).

We generated two response variables as means of comparison of the following null-hypothesis: 1) absence of differences in the explained variability (= variance or deviance) among the studies (Appendix S1). The variance and deviance were selected in order to compare the variability explained and, thus, the robustness of the results from all the analyzed studies [31,32], but taking into account that deviance is related to the number of studied variables and the sample size (Appendix S1), and 2) no differences in the frequency of a variable resulting statistically significant in the analyzed studies. We assumed that a variable might better explain the general selection patterns if being significant more times. For this later variable we also considered a) its positive/negative (+/−) relationship with the presence of cinereous vultures, and b) the proportion of times that each variable was significant in relation to the number of times it was studied.

Additionally, explanatory variables related to the data-sampling methods (*Sampling methods*, n = 4) and to aspects regarding the vulnerability and ecology of the species (*Ecological-vulnerability*, n = 3) were identified from the different studies. *Sampling methods* variables were intended for a better detailing of the meta-analysis and thus were included into the assessment of the effect of the methodological procedures used in each study on the explained variability. The variables considered were the type of sample considered in the analysis (point or polygon, *Sample*), the

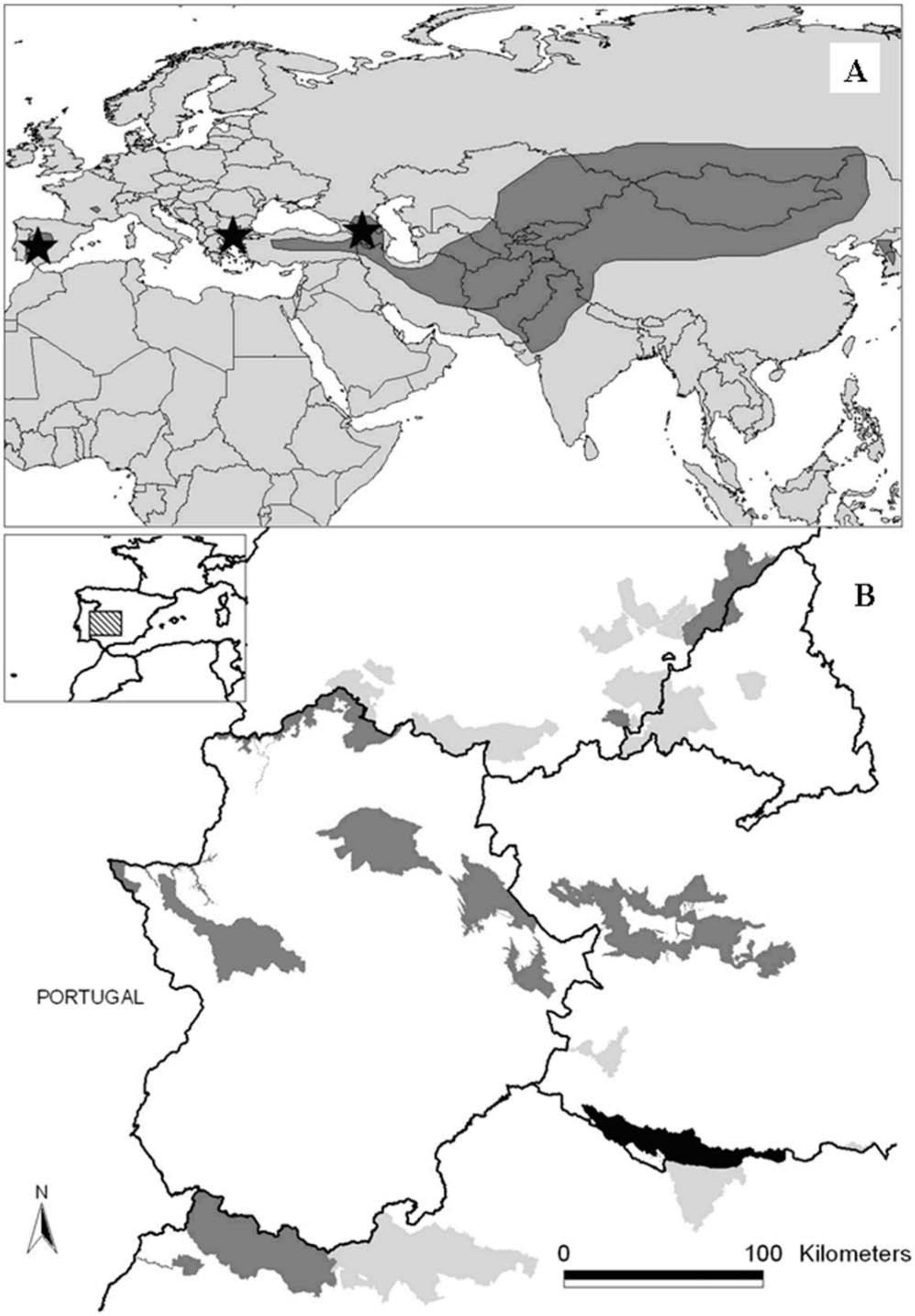

Figure 1. Global distribution range of cinereous vulture (dark grey) and metapopulations in which nest-site modelling studies and meta-analyses have been performed in this article (black stars). (A). Distribution in peninsular Spain of the Special Protected Areas (SPA) with presence of breeding cinereous vultures, specifying those in which nest-site modelling were studied (dark grey) and where the present field study was developed (black) (B).
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assignment to data set in only one year (yes or no, *Year of sampling*), the proportion between the number of nest-samples and number of random samples (*Nest/random*) and the location of random plots (1 = inside the perimeter of breeding colony; 2 = forest habitat in and around breeding colony; 3 = all habitats in and around the breeding colony, *Location of random samples*).

Studied breeding colonies were classified into *vulnerability* and *ecological* categories for a further analysis aiming at illustrating if any of the more significant kind of variables from those evaluated in the studies were related to the ecological and vulnerability characteristics of the studied populations. So, we identified as variables the vulnerability of the study area (1 = less than 40 pairs in the breeding colony and less than 500 pairs at national level; 2 = less than 40 pairs in the breeding colony and more than 500 pairs at national level; 3 = more than 40 pairs in the breeding colony and more than 500 pairs at national level, *Threat level*), the number of breeding pairs in the colony ($1 \leq 30$ pairs; $2 > 30$ pairs and ≤ 100 pairs; $3 > 100$ pairs, *Colony size*) and the type of vegetation (pine or oak, *Habitat*).

Statistical analyses

Present field study. To study the factors that affect nest-site selection, we selected a total of 240 points, that were analyzed at both microhabitat and landscape scales (*Microhabitat* and *landscape*). The analyses were performed with the software R.2.8.0 [33].

First, the variables to be included in each model were examined using Spearman's rank correlation (ρ) index to test the correlation between continuous candidate variables. Only non-correlated variables or those with weak correlation ($\rho < 0.3$) were included in the model carried out at each scale. We did not pose a multiple contrast hypothesis and subsequent selection using information criterion (AIC or similar) [34,35], in order to integrate the results of this study into the meta-analysis presented later in this work. All previous published papers on nesting-habitat selection of the cinereous vulture used the criteria of statistical significance, so we decided to maintain this criterion [13–16].

Response variables were binary (nest/random plot) and so we used generalised linear models with binomial family errors and logit-link functions. We looked for overdispersion using the dispersion parameter, which was calculated for each model by dividing the residual deviance by the residual degrees of freedom. Those models showing overdispersion were refitted by quasi-binomial family error [36].

The models were simplified by removing non-significant terms ($\alpha = 5\%$). Once we had determined the statistically significant factors in each model, we subsequently aggregated the non-significant levels of each factor to obtain the "minimal adequate model", by a stepwise *a posteriori* procedure. If two levels of a factor did not differ significantly and did not improve the fit of the model, they were grouped together [36].

Meta-analysis. First, we compared the variability explained by each of the studies, including the present field study results, for evaluating their ability to effectively model habitat selection from a methodological point of view. Hence, a meta-analysis testing the null-hypothesis of no-differences among the mean values of the deviance or variance was performed by using the Cochran's Q statistics of hetereogeneity. Previously, we checked that variance-deviance values of the studies fitted to a Chi-square distribution ($\chi 2 = 136.12$; df = 8; $p < 0.001$) [37]. I-squared test quantifies the degree of heterogeneity of the studied values by analysing the percentage of the whole observed differences in the deviance-variance values between studies which are not due to chance [38]. The effect-size of the studied variable (deviance or variance) and its confidence interval at 95% were also evaluated. ANOVA (for categorical variables) and regression (for continuous variables) analyses were performed to assess the influence of sampling procedures on the deviance-variance results. All analyses were performed with the software Comprehensive Meta-Analysis V.2 [20].

On the other hand, we analysed which variables from those studied predicted to be important in habitat selection. Thus, to evaluate the frequency of appearance of each statistically significant variable regarding nest-selection we chose variables that were significant most often by selecting those resulting significant more than once from the whole studies and provided that the variable was significant more than 1/3 of times it was considered. These variables were grouped according to their relationship with *microhabitat*, to the ecological characteristics at landscape level (*landscape*), to climate, to the effect of anthropic activities (*anthropic*) and others.

Subsequently, a Fisher's exact test was used to analyze the proportion of appearance of each type of significant variables (*landscape*, *microhabitat* and *anthropic*) and the characteristics of the studied population (in terms of the *threat level*, *colony size* and *habitat*). The software Statistica 6.1 [39] was used to perform these analyses.

Results

Habitat modelling from present field study

In relation to *microhabitat*, the variables *D_tree* and *H_tree* were highly correlated ($\rho = 0.58$) and we decided to include the first variable in the analysis, together with *Sp_tree* and their first-order interaction (Appendix S2). The levels 'cork oak' and 'juniper' of the factor *Sp_tree* were not apparently different from each other but were with respect to the level 'holm oak'. We checked that they could be combined into a single level without any statistically significant variation in the model by analysing changes in null deviance ($\chi^2 = -1.29$; df = 2; $p = 0.041$) to derive the minimal adequate model (Table 1). Vultures bred less often in holm oaks than in cork oaks and junipers. The interaction between species and tree diameter was also significant, and indicated that the diameter effect was greater in the case of holm oaks (see Table 1 for further details).

At *landscape* scale, we did not include in the model the variables *%_Qsuber* and *%_ othersp*, as they were correlated with *%_Qrot* ($\rho = -0.88$ and $\rho = -0.75$, respectively), nor *%_scree-rock* (correlated with *D_scree* $\rho = -0.61$) nor *Long_tracks* (correlated with *D_track* $\rho = -0.75$ and *D_road* $\rho = 0.36$). It was necessary to correct for overdispersion. South-facing slopes were selected for cinereous vulture to locate their nests in advance so northern, eastern and western orientation were joined obtaining a more parsimonious model from the precedent without statistical differences ($F = 0.91$, df = 2, $p = 0.40$) in order to simplify the number of levels of this variable. The model, after the simplification, it is shown in Table 1. A greater slope and the closeness of screes had a significant positive effect on nest-site selection; southern facing sites were selected in comparison with other orientations. Greater tree coverage within a radius of 100 m around the studied plot and a higher scrub layer showed a positive effect on nest presence. On the other hand, the presence of trees over 4 m high in a radius of 25 m around the nest as well as a greater cover of holm oak and shrub disfavoured selection by the cinereous vulture. Finally, the nests tended to be far from tracks, roads and human buildings (see Table 1 for further details).

Meta-analysis from published articles

In terms of sampling procedures, significant statistical differences in the deviance/variance existed between colonies ($Q = 89.18$, df = 12, $p < 0.001$, $I^2 = 86.54$, Figure 2). In addition, other factors also influenced the effect size of the deviance/variance reported in each of the analyzed studies: the location of the random plots ($Q = 52.81$, df = 2, $p < 0.001$, greater deviance/variance when plots are inside the perimeter of breeding colony), whether all data were sampled in a single year ($Q = 29.28$, df = 1, $p < 0.001$, greater deviance/variance when one-year sampling), the type of random data considered ($Q = 47.87$, df = 1, $p < 0.001$, greater deviance/variance for points) and the ratio between the number of nests sampled and the number of random samples ($Q = 89.18$, df = 1, $p = 0.002$, greater deviance/variance when lesser nest/random ratio).

The variables that were most often selected as statistically significant in nest-site selection for the whole Eurasian studies are presented in Table 2, being a greater slope, bigger diameter of the

Table 1. Dependence of nest-site selection on *microhabitat* characteristics and on variables at *landscape* scale from the selected minimal adequate models.

	Parameter	Effects±SE	z value	p value	Explained deviance
Microhabitat	Intercept	−3.7721±1.4289	−2.649	<0.001	0.18
	Sp.(cork oak+juniper) oak+juniper)	3.3882±1.6553	2.047	0.040	
	D_tree	0.1242±0.0397	3.127	<0.001	
	D_tree*Sp. oak+juniper	−0.0905±0.433	−2.088	0.036	
	Parameter	**Effects±SE**	**t value**	**p value**	**Explained deviance**
Landscape	Intercept	2.6212±5.9622	0.440	0.660	0.89
	Slope	0.5549±0.0954	5.813	<0.001	
	Orient_(N+E+W)	−4.0760±0.7230	−5.638	<0.001	
	D_scree	−0.0279±0.0052	−5.290	<0.001	
	Rad25_tree	−0.8642±0.1622	−5.327	<0.001	
	%tree	0.4772±0.08554	5.562	<0.001	
	H_shrub	3.3206±0.8554	3.882	<0.001	
	D_road	0.0018±0.0091	3.500	0.0011	
	D_track	0.0451±0.0090	4.875	<0.001	
	%_shrub	−0.0059±0.0155	−3.854	<0.001	
	Alt	−0.0564±0.0112	−5.014	<0.001	
	%_Qrot	−0.0601±0.0202	−2.971	0.003	
	D_const	0.0065±0.0019	6.043	<0.001	

Effects±SE were calculated considering the reference value of zero for Sp_Tree (olm oak) and the same for the interaction D_tree*Sp; Orient_(S).
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tree, the presence of scree around the nest-tree, orientation of the slope to south and a longer distance to tracks the five more proportionally and positively related to the presence of cinereous vulture nests. The number and type of significant variables regulating nest-site selection did not relate to the vulnerability of the colony (Table 3). Colony size conditioned the number of significant variables in relation to landscape such that in the largest colonies these variables were of greater importance (Table 3). In terms of habitat type, human activities had a greater negative effect in colonies in Mediterranean forests of *Quercus* sp. than in colonies located in pines (Table 3).

Discussion

Habitat selection

The present field study and the meta-analysis showed that the cinereous vulture selected nest-sites in large trees, on steep, south-facing slopes, close to screes and away from human infrastructures

Reference	Deviance/ Variance	Lower limit	Upper limit
[36] (1) Iruelas. SP	0,52	0,37	0,66
[36] (2) S. Pelada. SP	0,22	0,12	0,36
[34] Dadia. GR	0,89	0,72	0,96
[91] Rascafría. SP	0,96	0,88	0,98
[26] (1) Gata. SP	0,92	0,77	0,97
[26] (2) Granadilla. SP	0,91	0,76	0,96
[26] (3) Monfragüe. SP	0,94	0,79	0,98
[26] (4) Ibores. SP	0,76	0,60	0,87
[26] (5) S. Pedro. SP	0,84	0,68	0,92
[26] (6) Tajo. SP	0,90	0,75	0,96
[31] (1) Caucassus. GE	0,83	0,50	0,95
[31] (2) Caucassus. GE	0,84	0,51	0,96
Present study. Alcudia. SP	0,89	0,67	0,96
Total	0,75	0,70	0,80

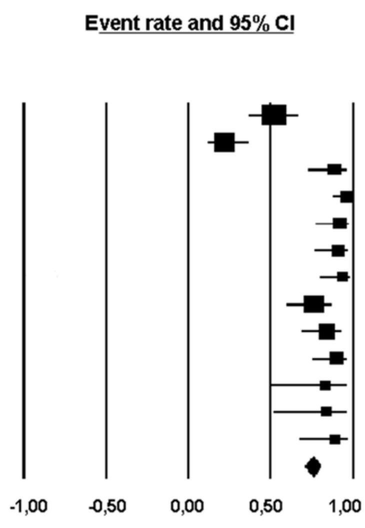

Figure 2. Effect size of the meta-analysis of the deviance-variance values of different studies on the breeding habitat selection of the cinereous vulture *Aegypius monachus*. The studies references, according to Appendix S1, its location (SP = Spain, GR = Greece, GE = Georgia), the values of deviance/variance and their confidence intervals at 95% are presented.
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Table 2. Most frequent significant variables resulting from nest-site selection in 16 cinereous vulture breeding colonies.

Variable	Type	Relation	n_{signif}	n_{stud}	Proportion
Slope	Landscape	+	8	12	0.67
Tree diameter	Microhabitat	+	4	6	0.67
Scree presence	Landscape	+	2	3	0.67
Orientation south	Landscape	+	3	5	0.60
Distance to track	Anthropic	+	7	12	0.58
Cork oak cover	Landscape	+	3	6	0.50
Human population index	Anthropic	−	3	6	0.50
Holm oak cover	Landscape	−	3	6	0.50
Position on upper third of the slope	Landscape	+	2	4	0.50
Tree height	Microhabitat	+	3	7	0.43
Distance to road	Anthropic	+	5	12	0.42
Orientation east	Landscape	+	3	5	0.60
Distance to nearest nest	Others	+	2	5	0.40
Altitude	Landscape	+	6	16	0.37

Relation indicates the type of relationship (positive or negative) of the variable with the selection by the cinereous vulture; (n_{signif}) shows the number of colonies for which the variables was statistically significant; (n_{stud}) indicates the number of colonies in which the variable was studied, and *Proportion* shows the ratio of the two previous numbers (n_1/n_2). The table includes variables that were statistically significant more than once and in more than one third of the studied colonies.

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or anthropogenic factors than can provoke disturbances. Steeper slopes determine nest-site selection in the cinereous vulture since disturbances are less likely to occur in rugged areas [40,41]. The largest trees cope better with the weight of nests and are easier to land on and take off from [16,17]. This robustness and height, often found around screes [42] and in old cork oaks, as well as the positively selected situation on the upper third of a hillside, ease the detection of predators and other sources of disturbance [12,43].

South- and east-facing slopes are selected as nest sites, probably because there are higher trees and better climate at a local scale [44]. Nonetheless, no climate-related variables were relevant in the general nest-site choice in the cinereous vulture, either due to the inter-annual heterogeneity of this feature, the fact that at local scale these variables do not predict adequately variations in ecological processes [45] or because not all the same variables were examined in the studies that we analyzed. Anthropic factors are very important in habitat selection, as occurs in other species that are wary of humans [46,47].

Nevertheless, the impact of human activities is a complex issue that should be case-by-case evaluated in relation to each breeding colony, since the secular management practices, the degree of habituation to human presence, the social awareness to this potential beneficial species or the ecological and biogeographical conditions may nuance its real influence [48,49]. In this sense, both vegetation and the availability and characteristics of trophic resources, as well as the impact of human activities, vary between regions approximately along a geographical gradient [50,51], although this species does have a certain plasticity in its ecological requirements [52]. These differences can be very marked as is shown by the fact that this species breeds on cliffs in Mongolia and Russia [53], thereby demonstrating that the factors affecting life histories in a single species with a wide range can be very heterogeneous [54,55].

Modelling and methodological conclusions

First, the number of studies analyzed ($n = 13$) was small to be able to draw definitive conclusions so patterns regarding the most efficient sampling approaches (see below) might be considered in relation to this low sample size. This is a common gap for meta-analysis works whose analytical procedures allow to integrate the onset of low sample sizes [32,37]. Nevertheless, this type of analysis of a single species has never previously been conducted in such a wide geographical context [56].

The total number of variables under study influences the results of the models and the variability that they reflect [57,58]. This implies that in habitat-modelling studies both the number and type

Table 3. P-values of the Fischer's exact test of the relation between variables that were significant in higher proportion in the different studies of nest-site selection of cinereous vulture (grouped in variables related to *microhabitat*, to *landscape* scale and to human interactions- *anthropic*-) to the threat level of the studied breeding colonies, to their colony size and to their type of habitat.

	Type of significant variable of the nesting-habitat selection		
Characteristics of the studied breeding colonies	*Microhabitat*	*Landscape*	*Anthropic*
Threat level	0.993	0.693	0.941
Colony size	0.561	0.015	0.552
Habitat	0.275	0.314	0.077

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of variables must be carefully chosen beforehand [59,60]. In this sense, it is worth highlighting the fact that it is important to choose the most explanatory variables and/or those that are easily repeatable [4]; otherwise, there is a risk that the processes will not provide information regarding the proposed objectives. This deficiency is especially relevant in the study of climate, since many different variables are used to study the same factors [e.g. temperature, humidity, rainfall, frost and wind) and provide only scattered and inconclusive information [30,61]. On the other hand, the influence of other factors such as the effects of global change, the use of integrating variables in ecological processes, diseases and certain biochemical factors [62,63] could modify the tendencies of the results obtained and could lead to the application of better planned and more efficient conservation policies [64]. Our study followed the patterns of choice of variables commonly considered in previous works, both in number and type, so it was not possible to include more interesting and complex analysis in the field work, which may reveal other significant factors.

The location of the random sampling points influences the variability detected in the study. Thus, if the random plots are situated within the colony the information obtained will be more detailed in terms of factors operating at a local scale. On the other hand, if random points are chosen at a scale that includes heterogeneous types of habitat, some of the more general variables are more likely to be significant [e.g. altitude, slope or vegetation cover; 13,57,65]. In our case we selected *a priori* the inner perimeter of the studied breeding colony as the framework for analysing differences between random and nests plots. One main objective was to assess differences in the selected habitat characteristics at precise level and thus, to show the sensitive factors for the breeding of cinereous vulture and to recommend the most suitable locations for developing land-use practices to local managers.

Our results reveal that if data are gathered during just one breeding season, the variability explained by the models increases, possibly because uncontrolled aspects such as inter-annual change regulating nest-site choice, weather conditions and individual behaviour are avoided [66]. Nevertheless, other studies have shown that bias could be reduced and variability in the results increased when long-term data incorporating temporary dynamics are analyzed [58,67]. Bias-variance trade-off determines general model fitting [58,66] and it is not possible to know exactly the bias integrated in each of the meta-analysed studies. Thus, our results should be interpreted with caution in this regard.

The election of points as random plots allows us to reflect greater deviance/variance in the model than if polygons are employed. Thus, results can throw light more accurately on questions regarding nest selection by species [16,68].

Lastly, it is important that the proportion between the sampled nests and the random plots is as balanced as possible, although if there is an unbalance it should be in favour of the random plots. In this way, when the relationship sample/random approaches 0, the explained variability increases [57,66,69]. Therefore, the election of a lesser number of random plots in relation to the nests in our field study could reduce the explained deviance (Table 1).

Implications for conservation

The results obtained show that habitat type and the size of breeding colony affect the type of variables that most influences nest-site selection. Thus, human activities have greater incidence in colonies situated in Mediterranean oak forests. This may be due to the relative ease of access to the colony, to the habitat quality or to the existence of the additional conservation problems [70,71]. In addition, it is possible that cinereous vultures may have a closer relationship with human activities in specific habitats such as pine forests, which have been exploited for a long time in a sustainable way with respect to the requirements of species present [72,73]. In Mediterranean oak forests, on the other hand, few human economic activities are undertaken during the breeding season (except cork harvesting) and so it is possible that in these environments the species is more sensitive to human presence [43].

Despite the existence of various forest management models [73], the results of our meta-analysis suggest the need to implement different generalized management policies in temperate forests of the Palaearctic: 1) Mature forests must be given priority in forest protection as they act as source of resources and diversity [48,72,74]. Our results showed that areas with greater trees and tree cover are the most valuable type of forests for the breeding cinereous vultures. 2) Economic activities often determine habitat selection by threatened species [75] and so exclusion areas should be established for the most threatening activities and/or agreements should be reached to make human activities compatible [12]. Cinereous vultures tend to locate their nests as far as possible from human presence so one of the management priorities should be the regulation of such activities [71]. 3) It is advisable to coordinate and to standardize the data sampling procedures in advance when planning habitat modelling studies for the same species at different geographical scales. It is thus interesting to make the effort of developing scientific and technical working groups integrated by experts and researchers dealing with species of conservation concern [6,74]. 4) The analysis of ecological processes that include variables that have not been taken into account to date in habitat modelling like those related to climate change, parasites-diseases or biochemical properties must be encouraged [64] and scientific evidence-based criteria must be applied on the basis of these specific studies [76].

According to these conclusions, the knowledge of habitat selection in indicator-endangered species is very valuable for optimizing evidence-based conservation actions [56,76,77]. Specifically, the modelling of species requirements should be undertaken for both conservation actions *ex situ* and *in situ*. Species reintroduction programmes should take into account the analysis of global patterns of habitat selection [78,79] and so studies evaluating ecological requirements are of great relevance for carrying out population viability analyses [80,81]. In the case of the cinereous vulture, it could be even more important given that one of the main conservation objectives for this species is the establishment of biological corridors that will connect currently isolated Palearctic populations [21,82] through reintroduction projects (e.g., in the Pyrenees, France, Balkan Peninsula).

Supporting Information

Appendix S1 Information contained in the articles consulted for the meta-analysis of the variation in the nest-site selection in the cinereous vulture *Aegypius monachus*. The variables considered for conducting the different analysis are shown (see *methods* for further information).
(DOC)

Appendix S2 Models evaluated for the analysis of nest-site selection by the cinereous vulture *Aegypius monachus* in the colony in Alcudia Natural Park, Spain. The scale of analysis, categorical (*factors included*) and continuous variables, the pairs of variables that were found to be correlated using the Spearman test (ρ), the interactions between pairs of variables included in the models, and the number of generalized linear models resulting that were analyzed are shown. Each model was built including all factors, non-correlated continuous variables and those with interactions with $\rho<0.30$, and only one of correlated variables each model.
(DOC)

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Author Contributions

Conceived and designed the experiments: RMO AA FG. Performed the experiments: RMO MFO AM AA FG. Analyzed the data: RMO MFO AM. Contributed reagents/materials/analysis tools: RMO MFO AM. Wrote the paper: RMO MFO AM AA FG.

References

1. Morrison ML, Marcot BG, Mannan RW (2006) Wildlife-Habitat Relationships. Concepts and Applications. 3rd Edition. Washington, DC: Island Press.
2. Margalida A, Donázar JA, Bustamante J, Hernández F, Romero-Pujante M (2008) Application of a predictive-model to detect long-term changes in nest-site selection in the Bearded Vulture *Gypaetus barbatus*: conservation in relation to territory shrinkage. Ibis 150: 242–249.
3. López-Bao JV, Palomares F, Rodríguez A, Delibes M (2010) Effects of food supplementation on home-range size, reproductive success, productivity and recruitment in a small population of Iberian lynx. Animal Conservation 13: 35–42.
4. Katzner T, Millner-Guland EJ, Bragin E (2007) Using modelling to improve monitoring of structured populations: Are we collecting the right data? Conservation Biology 21: 241–252.
5. Magana M, Alonso JC, Martín CA, Bautista LM, Martín B (2010) Nest-site selection by Great Bustards *Otis tarda* suggests a trade-off between concealment and visibility. Ibis 152: 77–89.
6. Pullin AS, Stewart GB (2006) Guidelines for systematic review in conservation and environmental management. Conservation Biology 20: 1647–1656.
7. Hoeksema JD, Forde SE (2008) A Meta-Analysis of Factors Affecting Local Adaptation between Interacting Species. The American Naturalist 171: 275–290.
8. Benítez-López A, Alkemade R, Verweij PA (2010) The impact of roads and other infraestructures on mammal and bird populations: a meta-analysis. Biological Conservation 143: 1307–1316.
9. Carrete M, Donázar JA (2005) Application of central-place foraging theory shows the importance of Mediterranean dehesas for the conservation of cinereous vulture *Aegypius monachus*. Biological Conservation 126: 582–590.
10. Şekercioğlu ÇH, Daily GC, Ehrlich PR (2004) Ecosystem consequences of bird declines. Proceedings of the National Academy of Sciences USA 101: 18042–18047.
11. Seddon PJ, Leech T (2008) Conservation short cut, or long and winding road? A critique of umbrella species criteria. Oryx 42: 240–245.
12. Margalida A, Moreno-Opo R, Arroyo BE, Arredondo A (2011) Reconciling the conservation of an endangered species with an economically important anthropogenic activity: interactions between cork exploitation and the Cinereous Vulture *Aegypius monachus* in Spain. Animal Conservation 14: 167–174.
13. Morán-López R, Sánchez JM, Costillo E, Villegas A (2005) Nest-site selection of endangered cinereous vulture *Aegypius monachus* populations affected by anthropogenic disturbance: present and future conservation implications. Animal Conservation 9: 29–37.
14. Gavashelishvili A, McGrady MJ, Javakhishvili Z (2006) Planning the conservation of the breeding population of cinereous vulture *Aegypius monachus* in the Republic of Georgia. Oryx 40: 1–8.
15. Poizaridis K, Goutner V, Skartsi T, Stamou G (2004) Modelling nesting habitat as a conservation tool for the Eurasian black vulture *Aegypius monachus* in Dadia Nature Reserve, northeastern Greece. Biological Conservation 118: 235–248.
16. Donázar JA, Blanco G, Hiraldo F, Soto-Largo E, Oria J (2002) Effects of forestry and other land-use practices on the conservation of cinereous vultures. Ecological Applications 12: 1445–1456.
17. Fargallo JA, Blanco G, Soto-Largo E (1998) Forest management effects on nesting habitat selected by european black vultures *Aegypius monachus* in central Spain. Journal of Raptor Research 32: 202–207.
18. Jiménez J (2002) Elaboración de un modelo de hábitat de reproducción de buitre negro *Aegypius monachus* en los Montes de Toledo. In: Torralvo C, ed. Anuario Ornitológico de Ciudad Real 1995–2001. Ciudad Real: SEO-Ciudad Real.
19. Cuevas JA, De la Puente J (2005) Hábitat potencial del buitre negro Aegypius monachus en la Sierra de Guadarrama. Serie Documentos n° 45. Madrid: CIA Fernando González Bernáldez.
20. Biostat (2006) Comprehensive Meta-Analysis. A computer program for meta-analysis. USA: Englewood.
21. Birdlife International (2008) *Aegypius monachus*. IUCN 2010. IUCN Red List of Threatened Species. Version 2010.1. Available from www.iucnredlist.com (accesed March 2011).
22. Donázar JA, Margalida A, Carrete M, Sánchez-Zapata JA (2009) Too sanitary for vultures. Science 326: 664.
23. Moreno-Opo R, Margalida A, Arredondo A, Guil F, Martín M, et al. (2010b) Factors influencing the presence of the cinereous vulture *Aegypius monachus* at carcasses: food preferences and implications for the management of supplementary feeding sites. Wildlife Biology 16: 25–34.
24. Simberloff D (1998) Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era? Biological Conservation 83: 247–257.
25. Palomares F, Delibes M, Ferreras P, Fedriani JM, Calzada J, et al. (2000) Iberian lynx in fragmented landscape: predispersal, dispersal and postdispersal habitats. Conservation Biology 14: 809–818.
26. González LM, Oria J, Sánchez R, Margalida A, Aranda A, et al. (2008) Status and habitat changes in the endangered Spanish Imperial Eagle *Aquila adalberti* population during 1974–2004: implications for its recovery. Bird Conservation International 18: 242–259.
27. De la Puente J, Moreno-Opo R, Del Moral JC (2007) El buitre negro en España. Censo nacional 2006. Madrid: SEO/BirdLife.
28. Morán-López R, Sánchez JM, Costillo E, Corbacho C, Villegas A (2006) Spatial variation in anthropic and natural factors regulating the breeding success of the cinereous vulture *Aegypius monachus* in SW Iberian Peninsula. Biological Conservation 130: 169–182.
29. Sergio F, Rizzolli F, Marchesi L, Pedrini P (2004) The importance of interspecific interactions for breeding-site selection: peregrine falcons seek proximity to raven nests. Ecography 27: 818–826.
30. Guisan A, Zimmermann NE (2000) Predictive habitat distribution models in ecology. Ecological Modelling 135: 147–186.
31. Williams DA (1987) Generalized Linear Model diagnostics using the deviance and single case deletions. Applied Statistics 36: 181–191.
32. Clarke GM, Morris AP (2010) A Comparison of Sample Size and Power in Case-Only Association Studies of Gene-Environment Interaction. American Journal of Epidemiology 171: 498–505.
33. R Development Core Team (2008) R: A language and environment for statistical computing. Vienna: R Foundation for Statistical Computing.
34. Burnham KP, Anderson DR (1998) Model Selection and Multimodel Inference. A Practical Information-theoretic Approach. New York: Springer.
35. Akaike H (1973) Information theory as an extension of the maximum likelihood principle. *Second International Symposium on Information Theory*. In Petrov BN, Csaki F, eds. Budapest: Akademiai Kiado.
36. Crawley MJ (2007) The R book. New York: John Wiley & Sons Ltd.
37. Kulinskaya E, Dollinger MB, Bjorkestol K (2011) Testing for Homogeneity in Meta-Analysis I. The One-Parameter Case: Standardized Mean Difference. *Biometrics* 67: 203–212.
38. Huedo-Medina TB, Sánchez-Meca J, Marín-Martínez F, Botella J (2006) Assessing heterogeneity in meta-analysis: Q statistic or I^2 index? Psychological Methods 1: 193–206.
39. StatSoft (2002) Statistica 6.1. Tulsa. Available from www.statsoft.com (accessed March 2011).
40. González LM, Arroyo BE, Margalida A, Oria J, Sánchez R (2006) Effect of human activities on behaviour and success of breeding Spanish imperial eagles *Aquila adalberti*. Animal Conservation 9: 85–93.
41. Aubad J, Aragon P, Rodríguez MA (2010) Human access and landscape structure effects on Andean forest bird richness. Acta Oecologica 36: 396–402.
42. Thompson JD (2005) Plant evolution in the Mediterranean. Oxford: Oxford University Press.
43. Blumstein DT, Fernández-Juricic E, Zollner PA, Garity SC (2005) Inter-specific variation of avian responses to human disturbance. Journal of Applied Ecology 42: 943–953.
44. Pérez MC (2007) Diversity in the cork oak forests of the northern Straits of Gibraltar Region (Southern Spain). Plant Ecology 189: 87–100.
45. Hallet TB, Coulson T, Pilkington JG, Clutton-Brock TH, Pemberton JM, et al. (2004) Why large-scale climate indices seem to predict ecological processes better than local weather. Nature 430: 71–75.
46. Sastre P, Ponce C, Palacín C, Martín CA, Alonso JC (2009) Disturbances to great bustards *Otis tarda* in central Spain: human activities, bird responses and management implications. European Journal of Wildlife Research 55: 425–432.
47. Moreno-Opo R, Fernández-Olalla M, Guil F, Arredondo A, Higuero R, et al. (2011) The role of ponds as feeding habitat for an umbrella species: best management practices for the black stork *Ciconia nigra* in Spain. Oryx 45: 448–455.
48. Begon M, Townsend CR, Harper JL (2006) Ecology: from individuals to ecosystems. 4th edition. Oxford: Blackwell Publishing.
49. Fisher J, Lindenmayer DB, Montague-Drake R (2008) The role of landscape texture in conservation biogeography: a case study on birds in south-eastern Australia. Diversity and Distributions 14: 38–46.
50. Doebeli M, Dieckmann U (2003) Speciation along environmental gradients. Nature 421: 259–264.

51. Mittelbach GG, et al. (2007) Evolution and the latitudinal diversity gradient: speciation, extinction and biogeography. Ecology Letters 10: 315–331.
52. Costillo E, Corbacho C, Morán R, Villegas A (2007) The diet of the black vulture *Aegypius monachus* in response to environmental changes in Extremadura (1970–2000). Ardeola 54: 197–204.
53. Nyambayar B, Bold A, Fuller M, Watson R (2006) Nesting ecology and breeding success of cinereous vultures *Aegypius monachus* in central Mongolia. Proceedings of 3rd International Ornithological Conference on Modern problems of ornithology in Siberia and Central Asia, Ulan-Ude.
54. Morrison C, Hero JM (2003) Geographic variation in life-history characteristics of amphibians: a review. Journal of Animal Ecology 72: 270–279.
55. Grotan V, Saether BE, Lillegard M, Solberg EJ, Engen S (2009) Geographical variation in the influence of density dependence and climate on the recruitment of Norwegian moose. Oecologia 161: 685–695.
56. Branton M, Richardson JS (2011) Assessing the value of the umbrella-species concept for conservation planning with meta-analysis. Conservation Biology 25: 9–20.
57. Fielding AH, Bell JF (1997) A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24: 38–49.
58. Drew CA, Wiersma Y, Huettman F (2010) Predictive species and habitat modelling in landscape ecology: concepts and applications. London: Springer.
59. Guisan A, Thuiller W (2005) Predicting species distribution: offering more than simple habitat models. Ecology Letters 8: 993–1009.
60. Jiménez-Valverde A, Lobo JM, Hortal J (2008) Not as good as they seem: the importance of concepts in species distribution modelling. Diversity and Distributions 14: 885–890.
61. Araujo MB, Guisan A (2006) Five (or so) challenges for species distribution modelling. Journal of Biogeography 33: 1677–1688.
62. Allen AP, Brown JH, Gillooly JF (2002) Global Biodiversity, Biochemical Kinetics and the Energetic-Equivalence Rule. Science 297: 1545–1548.
63. Hijmans RJ, Graham GH (2006) The ability of climate envelope models to predict the effect of climate change on species distributions. Global Change Biology 12: 2272–2281.
64. Austin M (2007) Species distribution models and ecological theory: A critical assessment and some possible new approaches. Ecological Modelling 200: 1–19.
65. Hirzel A, Guisan A (2002) Which is the optimal sampling strategy for habitat suitability modelling. Ecological Modelling 157: 331–341.
66. Scott JM, Heglund PJ, Morrison ML (2002) Predicting species occurrences: issues of accuracy and scales. Washington, DC: Island Press.
67. Brotons L, Herrando S, Pla M (2007) Updating bird species distribution at large spatial scales: applications of habitat modelling to data from long-term monitoring programs. Diversity and Distributions 13: 276–288.
68. Segurado P, Araujo MB (2004) An evaluation of methods for modelling species distributions. Journal of Biogeography 31: 1555–1568.
69. Engler R, Guisan A, Rechsteiner L (2004) An improved approach for predicting the distribution of rare and endangered species from occurrence and pseudo-absence data. Journal of Applied Ecology 41: 263–274.
70. McIntyre S, Hobbs R (1999) A framework for conceptualizing human effects on landscapes and its relevance to management and research models. Conservation Biology 13: 1282–1292.
71. Fernández-Juricic E, Jiménez MD, Lucas E (2001) Alert distance as an alternative measure of bird tolerance to human disturbance: implications for park design. Environmental Conservation 28: 263–269.
72. Lindenmayer DB, Margules CR, Botkin DB (2000) Indicators of biodiversity for ecologically sustainable forest management. Conservation Biology 14: 941–950.
73. Paillet Y, Bergès L, Hjältén J, Ódor P, Avon C, et al. (2009) Biodiversity differences between managed and unmanaged forests: Meta-analysis of species richness in Europe. Conservation Biology 24: 101–112.
74. Harris GM, Jenkins CN, Pimm SL (2005) Refining Biodiversity Conservation Priorities. Conservation Biology 19: 1957–1968.
75. Martínez-Abraín A, Oro D, Jiménez J, Stewart G, Pullin A (2010) A systematic review of the effects of recreational activities on nesting birds of prey. Basic and Applied Ecology 11: 312–319.
76. Sutherland WJ, Pullin AW, Dolman PM, Knight TM (2004) The need for evidence-based conservation. Trends in Ecology and Evolution 19: 305–308.
77. Olea PP, Mateo-Tomás P (2011) Spatially explicit estimation of occupancy, detection probability and survey effort needed to inform conservation planning. Diversity and Distributions 17: 714–724.
78. Seddon PJ, Armstrong DP, Maloney RF (2007) Developing the science of reintroduction biology. Conservation Biology 21: 303–312.
79. Armstrong DP, Seddon PJ (2008) Directions in reintroduction biology. Trends in Ecology and Evolution 23: 20–25.
80. IUCN (1998) Guidelines for Re-Introduction. IUCN/SSC Reintroductions Specialist Group. Gland-Cambridge: IUCN.
81. Sutherland WJ, Armstrong D, Butchart SHM, Earnhardt JM, Ewen J, et al. (2010) Standards for documenting and monitoring bird reintroduction projects. Conservation Letters 3: 229–235.
82. Bennet AF (2003) Linkages in the Landscape: The Role of Corridors and Connectivity in Wildlife Conservation. Gland-Cambridge: IUCN.

CAPÍTULO-CHAPTER
2

Influence of environmental factors on cinereous vulture breeding success in central Spain

by

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Factores ambientales que determinan el éxito reproductivo del buitre negro en el centro de España

El éxito reproductivo es un parámetro demográfico clave que condiciona la dinámica de las poblaciones de aves y, por tanto, la comprensión de los factores que lo determinan resulta importante para la aplicación de medidas de conservación de especies amenazadas. En este sentido, el éxito reproductivo depende de distintas variables, no excluyentes entre sí, como son las estrategias vitales desarrolladas por las especies así como por variables ambientales, ecológicas y antrópicas. Tomando como especie modelo el buitre negro, se evaluó la influencia sobre el éxito reproductivo de las características del nido, del árbol que albergó el nido, de la composición paisajística y de molestias antrópicas, en una colonia de cría del centro de España. Para hacerlo, se seleccionaron y estudiaron variables relativas a la vegetación, la geomorfología y la presencia humana. El éxito reproductivo resultó influido significativa y positivamente por un mejor estado de conservación de las plataformas de nidificación, una mayor altura de los árboles-nido y una situación de éstos en pendientes más pronunciadas, orientadas al sur y con una elevada cobertura de alcornoque en un radio de 100 m. Este patrón reflejaría la influencia que tienen sobre las probabilidades de éxito las condiciones climatológicas, de accesibilidad a los nidos o de la propia selección de hábitat de nidificación. Las conclusiones extraídas están condicionadas por la metodología de trabajo empleada y ponen de manifiesto la necesidad de homogeneizarlas en estudios comparados. Se recomienda que los bosques maduros en áreas poco accesibles y en laderas de elevada pendiente sean gestionados convenientemente y que estas masas forestales sean protegidas de molestias humanas con el propósito de promover la conservación de áreas de reproducción de rapaces amenazadas de medios forestales.

ABSTRACT

Breeding success is a key demographic parameter determining the fate of bird populations and, therefore, understanding its determinants is an important issue for the application of conservation measures in the case of endangered species. In this sense, breeding success depends on diverse, not mutually exclusive, effects such as the life strategies developed by species, as well as environmental, ecological and anthropic factors. The influence of nest and nest tree characteristics, landscape composition and human disturbance factors on the breeding success of a Cinereous Vulture *Aegypius monachus* breeding colony in central Spain was analysed. To do so, a number of variables on the basis of the characteristics of the existing vegetation around the nest-tree and the own features of the nest and the nest-tree were selected. Our results suggest that better-conserved and bigger nesting platforms, taller nesting trees and a higher tree and shrub coverage in a radius of 100 m around the nest resulted in an increased success of the breeding pairs. These findings shows the influence that nest orientation and location, nest accessibility and habitat selection have on breeding success, also stressing the importance to take into account similar studies for the design of management strategies. We recommend that mature forests in less-accessible, steeper slopes are carefully conserved and that these forests are protected from anthropogenic disturbances in order to strengthen the conservation of breeding areas of forest-dwelling endangered raptor species.

KEY WORDS

Aegypius monachus; forest management; vegetation influence; conservation practice; threatening factors; human disturbance; landscape features.

INTRODUCTION

Birds exhibit a number of diverse life-traits with large, long-lived species typically showing low fecundity and high adult survival rates (Cody 1971, Bdayaev & Ghalambor 2001). This is more evident in many birds of prey with highly specific nutritional and habitat requirements (Newton 1979) whose life strategies are influenced by environmental factors (Erikstad *et al*. 1998). The study and knowledge of the inherent details of species' life-traits is essential if suitable management measures are to be applied, mainly when their populations are threatened (Pullin *et al*. 2004, Oro *et al*. 2008).

The trade-off between survival and reproduction is a key aspect in the understanding of birds' life strategies (Linden & Møller 1989). Thus, for a long-lived species it is more important in terms of its overall fitness to attain high adult survivorship than to rapidly engender many offspring (Hirshfield & Tinkle 1975, Erikstad *et al*. 1998). This means that the most important task in the successful resolution of conservation questions concerning large threatened birds is the reduction of non-natural adult mortality (Oro *et al*. 2008, Ortega *et al*. 2008, Grande *et al*. 2009). Nevertheless, this principle must be implemented within a realistic framework and thus take into account the real possibilities that management action will be effective (McCarthy & Possingham 2007). So, other demographic parameters, such as the reproduction rates, might be dealt to favour the conservation status in the case of endangered species.

Breeding success in birds, i.e. the number of chicks fledged in each female's breeding attempt, is dependent on a wide-range of factors. For instance, the survival of newly hatched chicks is affected by climatic conditions (Kostrzewa & Kostrzewa 1991, Margalida *et al*. 2007), as well as by landscape features, the possibilities of finding food and the ability to ward off predators (Lind & Creswell 2005, Wilkin *et al*. 2009). As well, human presence, which generally involves activities that are likely to have a negative impact on the breeding attempt, affects the number of chicks fledged, even if the disturbance occurs once the breeding cycle has begun (González *et al*. 2006, Zuberogoitia *et al*. 2008, Margalida *et al*. 2011). Disturbances not only influence breeding success during a breeding attempt, but can also lead to changes in distribution patterns and, even, changes in individual behaviour (Sutherland 2007).

The recovery programmes for threatened species implies to understand both the choice of breeding habitat and the factors that determine breeding success (Lindenmayer *et al*. 2008). An abundant body of literate exists on threatened raptors and habitat selection (i.e. González *et al*. 1992, Margalida *et al*. 2008, Bosch *et al*. 2010), although information on identifying the factors that determine the success or failure of the breeding attempt is somewhat scarcer (Jenking 2000, Arroyo & Razin 2006, Bionda & Brambilla 2012). In this study we evaluated the influence of environmental factors and human-related activities on the breeding success of Cinereous Vulture *Aegypius monachus*, in order to understand which factors are the most relevant and thus be able to take them into account in the management of the areas this raptor inhabits. In this sense, this vulture is one of the species that has received most attention internationally from conservation programmes (Heredia 1996, BirdLife International 2008) and so the long-term effects of this active territorial management can be evaluated and the efficiency of the measures carried out can be studied in as much detail as is necessary (Moreno-Opo & Guil 2007).

METHODS

The study species

The Cinereous Vulture is the largest Palearctic raptor and is considered to be Near Threatened (7,200-10,000 pairs, BirdLife International 2008). In Spain a total of 1,825 pairs occur (De la Puente *et al*. 2007) and it is regarded as Vulnerable (Ministry of Environment, Rural and Marine Affairs 2011). Its habitat is to be found in upland areas in the south-west of the Iberian Peninsula and in the island of Mallorca. It breeds colonially in forests and builds large nests on treetops (Donázar 1993). It reaches the sexual maturity at 5-6 years of age (Cramp 1998), although the earliest successful breeding attempts have been registered at 3 years old (Terrasse *et al*. 2004). Nests occupied by different pairs are located at varying distances (average 556.6 ± 606.9 m, Morán-López *et al*. 2006a), while the same pairs usually hold an average of 2.4 nests (De la Puente 2007) in a different state of repair within a shorter distance each other. Only one chick per breeding attempt fledges (Hiraldo 1983). It is one of the bird species with the longest breeding cycle throughout the Palearctic (Cramp 1998); in Spain, the mating behaviour begins in late December and lasts until February, laying eggs between February and April. After about 59 days of incubation (range 51-68) chicks hatch, and they will not start their first flights tries until 114 ± 11 days, which usually occurs during August and September (Del Moral & De la Puente 2005). Its feeding behaviour determines both its use of space and the time invested in different activities. As a scavenger species it feeds on the carcasses and remains of various animals, mainly domestic cattle and populations of wild lagomorphs and ungulates (Costillo *et al*. 2007).

Study area

Fieldwork was carried out in the breeding colony of Umbría de Alcudia, Castilla-La Mancha, central Spain, the fourth largest in Spain (De la Puente *et al*. 2007). This site is protected as a Special Protected Area (SPA) for birds and consists of an upland area at altitudes of 736-1,115 m a.s.l. The vegetation on the slopes of the nesting colony is dominated by typical Mediterranean trees such as holm oak (*Quercus rotundifolia*), cork oak (*Quercus suber*), strawberry-tree (*Arbutus unedo*), prickly juniper (*Juniperus oxycedrus*), Lusitanian oak (*Quercus faginea*) and, to a lesser extent, associated shrubs in areas with a developed shrub layer.

Field work and studied variables

Fieldwork was carried out during October 2005 to January 2006. During 2005 breeding season, a monitoring program was developed to check occupation and success of the Cinereous Vulture. In order to reduce disturbances from the field technicians, after breeding season the occupied nests (n = 89) were visited for the measurement of the variables. The nests in which a chick was fledged (n = 50) were compared to occupied nests that failed to fledge a young bird (n = 39).

Independent variables of the study were selected on the strength of either their possible influence on the species' breeding success as reported in previous studies (Donázar et al. 2002, Morán-López *et al*. 2006b), or as factors related to the management of the selected habitats, as per modelling tendencies in raptor habitats (Poizaridis *et al.* 2004). Five types of variables were taken into account: tree and nest characteristics, geomorphologic variables, variables related to the vegetation, and variables linked to human disturbance (n = 20 total variables, Table 1). Climatic variables were considered to be irrelevant to breeding success given the relatively small surface area of the study site (total 11,115 ha) and were not included in the analyses. The success or failure of the breeding attempt was chosen as the study's response variable (Guisan & Zimmerman 2000).

Table 1. Independent variables considered in the analysis of the characteristics that determine the breeding success of the Cinereous Vulture *Aegypius monachus* in the colony in Umbría de Alcudia, Ciudad Real, Spain. *continuous variable, **categorical variable.

Variable	Description	Study level
*Sp_tree***	Tree species	*Tree characteristics*
*H_tree**	Total height of nest tree m	
*Nest_cons***	Conservation status of the nest: "good" = well structured, with upper herbaceous cover, circular or squared shaped, "not good" = with any imperfection, with upper herbaceous cover but not circular or squared shaped, "bad" = old, not structured nor renewed, without upper herbaceous cover, partially collapsed or fallen	*Nest characteristics*
*D_nest**	Length of the longer diameter of the platform of the nest cm	
*Alt**	Altitude m asl	*Geomorphologic*
*Scree***	Presence of nest/random tree in a natural scree "yes"/"no"	
*Orient***	Orientation where nest/random tree is located "N", "S", "W", "E"	
*D_scree**	Distance from nest/random tree to nearest natural scree m	
*Rad25_tree**	Number of tree higher than 4m existing around the nest tree in a 25m radius	*Vegetation*
*H_shrub**	Medium high of the shrub in a 100m radius to nest tree m	
*%_tree**	Percentage of tree coverage in a 100m radius around nest tree %	
*%_shrub**	Percentage of shrub coverage in a 100m radius around nest tree %	
*%_scree-rock**	Percentage of scree or rock outcrop coverage in a 100m radius around nest tree %	
*%_Qsuber**	Coverage of cork oak *Quercus suber* in tree covered surfaces in a 100m radius around the nest tree %	
*%_Qrot**	Coverage of holm oak *Quercus rotundifolia* in tree covered surfaces in a 100m radius around the nest tree %	
*%_othersp**	Coverage of other tree species in tree covered surfaces in a 100m radius around the nest tree %	
*Long_tracks**	Length of not paved tracks in a 500m radius around the nest tree m	*Human disturbance*
*D_road**	Distance from the nest tree to the nearest paved road m	
*D_const**	Distance from the nest tree to the nearest human building m	
*D_track**	Distance from the nest tree to the nearest unpaved track m	

Statistical analyses

To select the independent variables to be included in each model the correlation between the continuous candidate variables was checked using Spearman´s rank correlation (ϱ) index. The variable *%_Qrot* (see Table 1) showed strong correlations with *%_scree-rock* (ϱ= -0.61) and *%_othersp* (ϱ= -0.66) and, therefore, *%_Qrot* was not included in any model. In order to explore the influence of each group of variables in Cinereous Vulture breeding success, we established a set of *a priori* competing generalized lineal models (6 models, see Table 2). In all cases, the response variables were binary (success/failure) and so we used binomial family errors and logit-link functions. We looked for overdispersion through the dispersion parameter, which was calculated for each model by dividing the residual deviance by the residual degrees of freedom (Crawley 2007). No model showed dispersion parameters greater than 1.5, so we did not have to compensate for overdispersion

(Crawley 2007). Model selection was done using Akaike's Information Criteria (AIC), duly corrected for small sample size, as well as Akaike's weights (wj, for each model j) as an index of the strength of evidence of each model (Burnham & Anderson 2002). All analyses were conducted with the software R.2.12.0 (R Development Core Team 2008).

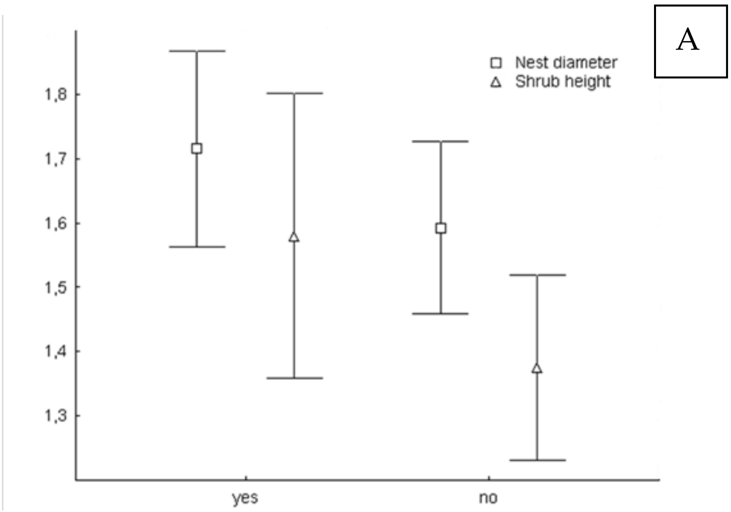
RESULTS

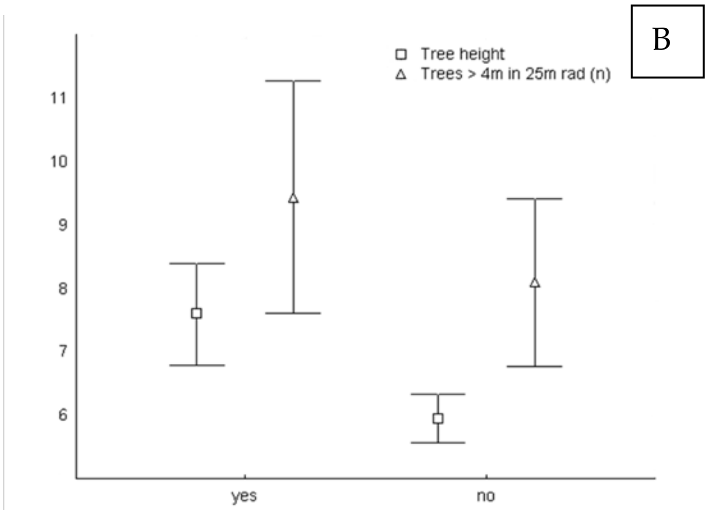
After comparing and selecting the models exploring the effects in Cinereous Vulture breeding success, the most parsimonious model (model 6) included the effect of nest and tree characteristics, together with variables describing vegetation around each nest (Table 2).

From the model 6, we found that breeding success increases according to the height of the nest-tree (0.43 –coefficient- ± 0.15 -standard error-), the diameter of the nesting platform (0.13 ± 0.52), the abundance of trees higher than 4m around the nest-tree (0.11 ± 0.05), the height of the shrub (0.73 ± 0.41) and the coverage of tree (0.06 ± 0.02 for all tree species, 0.01 ± 0.01 for cork oak, 0.11 ± 0.05 for other species than cork and holm oak), shrub (0.04 ± 0.01) and scree (0.02 ± 0.01) within 100 m around the nest (Figure 1). By contrast, breeding success seemed to be negatively affected by the degree of deterioration of the nest (-2.43 ± 1.51).

Table 2. Multiple hypotheses testing aimed at assessing the effect of Tree, Nest, Vegetation and Geomorphologic characteristics on Cinereous Vulture *Aegypius monachus* breeding success, as well as the effect of human-related disturbance. All the variables considered in these characteristics see Table 1 are included in the models. K: number of variables, AICc: Akaike's Information Criterion corrected for small sample size, w_i: Akaike weight of each model.

Model	**Deviance**	**K**	**AICc**	**ΔAICc**	**w_i**	**% explained deviance**
Model 1: Tree characteristics + Nest characteristics + Vegetation + Geomorphologic + Human disturbance	113.60	20	104.44	3.87	0.09	32.29
Model 2: Tree characteristics + Nest characteristics	114.73	4	115.73	15.15	0.00	4.98
Model 3: Geomorphologic	111.54	4	121.45	20.88	0.00	1.39
Model 4: Vegetation	118.66	8	114.00	13.43	0.00	9.59
Model 5: Human disturbance	118.66	4	122.24	21.67	0.00	0.67
Model 6: Tree characteristics + Nest characteristics + Vegetation	114.73	12	100.57	0.00	0.62	24.84





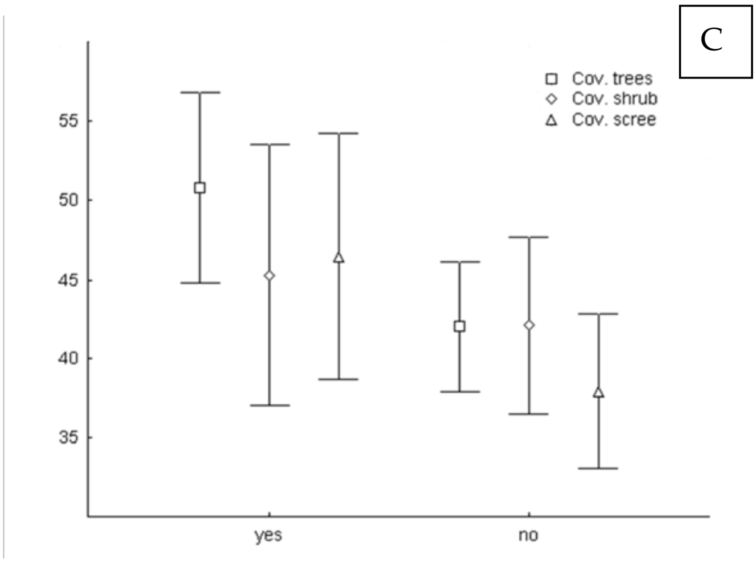


Figure 1. Mean values ± 95% confidence interval of the variables included in the most parsimonious model (model 6, Table 2) in relation to the breeding success yes/no. (A) Nest diameter and shrub height in m. (B) Tree height in m and number (n) of trees taller than 4m height within a radius of 25m around the nest. (C) Coverage in % of trees, shrub and scree around a radius of 100m around the nest.

DISCUSSION

The results obtained in this study show how variables related to the solidity of the nest structure, to the height of the nest-tree, to the length of the diameter of the nest and to the vegetation at landscape scale (greater tree, shrub and scree coverage and high of the shrub) may influence the breeding success of the Cinereous Vulture. Firstly, an adequate support where to place the nest is an essential precondition for viable incubation and chick-rearing periods (Margalida & Bertran 2000, Dalley *et al*. 2008). For this reason, breeding vultures constantly furnish the nest with additional material to ensure it is always in a good state of repair. Taller nesting trees facilitate the tasks involved in successful nesting: they make it easier for birds to access the nest (Donázar *et al.* 2002), increase the possibility of detecting predators or other sources of disturbance (Magana *et al*. 2010, Margalida *et al*. 2011) and can support the weight of larger nests (Fargallo *et al*. 1998). At vegetation level, the determinant variables in breeding success could be related to 1) accessibility: dense forests with high shrub hamper access by pedestrian potential predators or other elements liable to cause disturbance (Aubad *et al*. 2010), and 2) a positive selection of certain plant communities, in this case, cork-oak forests or prickly juniper (authors unpubl. data).

The results from present work differ from those reported by previous studies that have evaluated factors determining reproductive success of the Cinereous Vulture in other Spanish colonies (Donázar *et al*. 2002, Morán-López *et al*. 2006b). First of all, it has to be taken into account that the present work has been achieved for one breeding season, so conclusions might be considered within these circumstances (Brotons *et al.* 2007). In terms of the variables at landscape scale, in all previous studies greater breeding success was found to occur in areas at higher relative altitude, on steeper, more rugged slopes, and on slopes not facing north (Donázar *et al*. 2002, Morán-López *et al*. 2006b). Other important issue when managing Cinereous Vulture breeding territories is the effect of human disturbance on breeding success (Margalida *et al*. 2011). Variables related to this effect were only reported to influence breeding success in the two colonies studied by Donázar *et al*. (2002). They found that less human presence had a positive effect on breeding success. This factor is generally seen to be important on nesting habitat selection in the Cinereous Vulture (Fargallo *et al*. 1998, Poizaridis *et al*. 2004, Gavashelishvili *et al*. 2006, Morán-López *et al*. 2006a). Moreover, previous studies on Cinereous Vulture´s breeding success considered other types of variables, such as climatic ones, obtaining that higher temperatures, less rain during the time the chicks are in the nest and more rain in the preceding winter favour breeding success (Morán-López *et al.* 2006b). On the other hand, before the current study no previous work had ever analyzed the importance of variables related to the micro-habitat scale (nest and tree characteristics).

One important variable determining the breeding success of the Cinereous Vulture not taken into account in these studies, including the present one, is the quality of the breeding birds (Robinson *et al*. 2005, Sanz-Aguilar *et al*. 2008). There is a correlation between the physical characteristics of the individual (i.e. age, experience, etc.) and the quality of the breeding site, as occurs in other bird species (Velando & Freire 2001). Additionally, the quality of the breeding site may also be related to the birds´ preferences or to the direct effects of environmental characteristics on the breeding success (Lascroel *et al*. 2009). The consideration of these factors in the study might have increased the accuracy and representativeness of the results, but we did not achieve a priori the assessment of the breeding quality of the birds.

The use of heterogeneous modelling procedures could affect the precision of results generated by comparing different studies (Katzner *et al*. 2007, Drew *et al*. 2010). Besides differences in the variables studied, sources of variation among studies also exists in a number of other aspects (Moreno-Opo *et al.* 2012): the type of habitat of the studied colonies (nests in pines or oaks), the number of studied breeding pairs (Donázar *et al*. 2002, Morán-López *et al*. 2006b), ecological and socio-economic differences between different areas, data gathering carried out in just one or in more than one years (Brotons et al. 2007) or the number of variables studied (Moreno-Opo *et al.* 2012). Moreover, numerous non-controlled variables could contribute to determine the breeding success in this species, which may influence the results, as the values of explained variance or deviance would seem to suggest for the studied colonies in Donázar *et al.* (2002) (0.02, 0.03), Morán-López *et al.* (2006b) (0.15, 0.59, 0.63) and the present study (0.24). The non-controlled variables could be a product of physiological, health or genetic aspects of the birds themselves (Ricklefs & Wikelski 2002, Hogstad 2005), or mortality factors that affect certain individuals according their age class or sex (Hernández & Margalida 2008, Margalida *et al*. 2008).

Implications for the conservation of endangered forest species

Firstly, mature, well-evolved forests harbouring tall trees must be conserved. Such forest are the source areas and guarantee good numbers of offspring that will increase the relative abundance of the species (Lindenmayer *et al*. 2000) and mitigate the effects of non-controlled mortality (Hernández & Margalida 2008). Secondly, areas of mature forest in wild landscapes with steep slopes are the most critical breeding areas for large forest raptors such as the Cinereous Vulture. Finally, human presence and activity in natural areas negatively affect the presence of threatened species (Martínez-Abraín *et al*. 2010) and their reproductive success (Donázar *et al*. 2002). Thus, it is essential to establish buffer areas to avoid disturbance by forestry management (Arroyo & Razin 2006, González *et al*. 2006) and/or reach collaboration agreements with local agents to ensure that management tasks are compatible with conservation aims (Margalida *et al*. 2011).

Nevertheless, these recommendations should not eschew the aspects that most affect the conservation of a long-lived raptor species: 1) the need to guarantee a suitable survival rate in adult birds (Oro *et al.* 2008, Carrete *et al.* 2009, Ortega *et al.* 2009) by fighting the illegal use of poisoned bait and accidents occurring on certain types of large infrastructures (Guitart *et al.* 2010), and 2) the maintenance of a minimum surface area of habitat in good conservation status in which birds can carry out their breeding, feeding and rest requirements (Harris *et al*. 2005).

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REFERENCES

Akaike H. 1973. Information theory as an extension of the maximum likelihood principle. In: Petrov B. N. & Csaki F. (eds) Second International Symposium on Information Theory, Akademiai Kiado, Budapest, pp. 267–281.

Arroyo B. & Razin M.. 2006. Effect of human activities on bearded vulture behaviour and breeding success in the French Pyrenees. Biological Conservation 128: 276-284.

Aubad J., Aragon P. & Rodríguez M. A. 2010. Human access and landscape structure effects on Andean forest bird richness. Acta Oecologica 36: 396-402.

Badyaev A.A. & Ghalambor C.K. 2001. Evolution of life histories along elevational gradients: trade-off between parental care and fecundity. Ecology 82:2948–2960.

Bionda R. & Brambilla M. 2012. Rainfall and landscape features affect productivity in an alpine population of Eagle Owl *Bubo bubo*. Journal of Ornithology 153: 167-171.

Birdlife International. 2008. *Aegypius monachus*. In: IUCN 2010. IUCN Red List of Threatened Species, Version 2010.1. www.iucnredlist.org.

Bosch R., Real J., Tintó A., Zozaya E. L. & Castell C. 2010. Home-ranges and patterns of spatial use in territorial Bonelli's Eagles *Aquila fascista*. Ibis 152:105-117.

Brotons L., Herrando S. & Pla M. 2007. Updating bird species distribution at large spatial scales: applications of habitat modelling to data from long-term monitoring programs. Diversity and Distributions 13:276-288.

Burnham K.P. & Anderson D.R.1998. Model Selection and Inference. A Practical Information-theoretic Approach, Springer-Verlag, New York.

Carrete M., Sánchez-Zapata J.A., Benítez J. R., Lobón M. & Donázar J.A. 2009. Large scale risk assessment of wind-farms on population viability of a globally endangered raptor. Biological Conservation 142:2954-2961.

Cody M.L. 1971. Ecological aspects of reproduction. In: Farner D.S. & King J.R. (eds) Avian Biology 1, Academic Press, New York, pp. 461-512.

Costillo E., Corbacho C., Morán R. & Villegas A. 2007. The diet of the black vulture *Aegypius monachus* in response to environmental changes in Extremadura 1970-2000. Ardeola 54:197-204.

Cramp S. 1998. The Birds of the Western Palearctic on CD-Rom, Oxford University Press, Oxford.

Crawley M.J. 2007. The R book, John Wiley and Sons Ltd., New York.

Dalley K.L., Taylor P.D. & Shutler D. 2008. Nest-site characteristics and breeding success of three species of boreal songbirds in western Newfoundland, Canada. Canadian Journal of Zoology 86:1203-1211.

De la Puente J. 2007. Nesting-habitat selection of the cinereous vulture in pine forests. In: Moreno-Opo, R. & Guil F. (eds) Handbook on habitat and populations management of cinereous vulture in Spain, Ministry of Environment, Madrid pp.53-57.

De la Puente J., Moreno-Opo R. & Del Moral J. C. 2007. El buitre negro en España. Censo nacional 2006, SEO/BirdLife, Madrid.

Del Moral, J.C. and J. De la Puente. 2005. Cinereous vulture *Aegypius monachus*. *In* Salvador A and L. M. Carrascal [eds] Enciclopedia virtual de los vertebrados ibéricos, http://www.vertebradosibericos.org/aves/aegmon.html, Accessed on 9th November 2011.

Donázar J.A. 1993. Los buitres ibéricos, Biología y Conservación, J. M. Reyero Editor, Madrid.

Donázar J.A., Blanco G., Hiraldo F., Soto-Largo E. & Oria J. 2002. Effects of forestry and other land-use practices on the conservation of cinereous vultures. Ecological Applications 12:1445-1456.

Drew C.A., Wiersma Y. & Huettman F. 2010. Predictive species and habitat modelling in landscape ecology: concepts and applications, Springer, London.

Erikstad K.E., Fauchal P., Tveraa T. & Steen H. 1998. On the cost of reproduction in long-lived birds: the influence of environmental variability. Ecology 79:1781–1788

Fargallo J.A., Blanco G. & Soto-Largo E.1998. Forest management effects on nesting habitat selected by eurasian black vultures *Aegypius monachus* in central Spain. Journal of Raptor Research 32:202-207.

Fernandez C., Azkona P. & Donázar J.A. 1998. Density-dependent effects on productivity in the Griffon vulture *Gyps fulvus*: the role of interference and habitat heterogeneity. Ibis 140:64-69.

Gavashelishvili A., McGrady M.J. & Javakhishvili Z. 2006. Planning the conservation of the breeding population of cinereous vulture *Aegypius monachus* in the Republic of Georgia. Oryx 40:1-8.

González L.M., Bustamante J. & Hiraldo F. 1992. Nesting habitat selection by the Spanish imperial eagles *Aquila adalberti*. Biological Conservation 59: 45-50.

González L.M., Arroyo B.E., Margalida A., Oria J. & Sánchez R. 2006. Effect of human activities on behaviour and success of breeding Spanish imperial eagle *Aquila adalberti*. Animal Conservation 9:85-93.

Grande J.M., Serrano D., Tavecchia G., Carrete M., Ceballos O., Díaz-Delgado R., Tella J.L. & Donázar J.A. 2009. Survival in a long-lived territorial migrant: effects of life-history traits and ecological conditions in wintering and breeding areas. Oikos 118:580-590.

Guisan A. & Zimmermann N.E.. 2000. Predictive habitat distribution models in ecology. Ecological Modelling 135:147–186.

Guitart R., Sachana M., Caloni F., Croubels S., Vandenbroucke V. & Berny P. 2010. Animal poisoning in Europe. Part 3. Wildlife Veterinarian Journal 183:260–265.

Harris G.M., Jenkins C.N. & Pimm S.L. 2005. Refining biodiversity conservation priorities. Conservation Biology 19: 1957-1968.

Heredia B. (ed) 1996 Action plan for the Cinereous Vulture *Aegypius monachus* in Europe, BirdLife International-European Commission.

Hernández M. & Margalida A. 2008. Pesticide abuse in Europe: effects on the cinereous vulture *Aegypius monachus* population in Spain. Ecotoxicology 17: 264-272.

Hirshfield M F. & Tinkle D.W. 1975. Natural selection and the evolution of reproductive effort. Proceedings of the National Academy of Sciences USA 72: 2227-2231.

Hogstad O. 2005. Sex-differences in nest defence in Fieldfares *Turdus pilaris* in relation to their size and physical condition. Ibis 147: 375-380.

Jenking A.R. 2000. Factors affecting breeding success of Peregrine and Lanner Falcons in South Africa. Ostrich 71: 385-392.

Katzner T., Millner-Guland E.J. & Bragin E. 2007. Using modelling to improve monitoring of structured populations: are we collecting the right data? Conservation Biology 21: 241-252.

Kostrzewa R. & Kostrzewa A.1991. Winter weather, spring and summer density, and subsequent breeding success or Eurasian kestrels, common buzzards, and northern goshawk. The Auk 108: 342-347.

Lascroel A., Dugger K.M., Ballard G. & Ainly D.G.. 2009. Effects of individual quality, reproductive success and environmental variability on survival of a long-lived seabird. Journal of Animal Ecology 78: 798-806.

Lind J. & Creswell W. 2005. Determining the fitness consequences of antipredation behavior. Behavioural Ecology 16: 945-956.

Linden M. & Møller A.P. 1989. Cost of reproduction and covariation of life history traits in birds. Trends in Ecology and Evolution 4: 367-371.

Lindenmayer D.B., Margules, C.R. & Botkin D.B.. 2000. Indicators of biodiversity for ecologically sustainable forest management. Conservation Biology 14: 941-950.

Lindenmayer D. *et al*. 2008. A checklist for ecological management of landscapes for conservation. Ecology Letters 11:.78–91.

McCarthy M.A. & Possingham H.P. 2007. Active Adaptive Management for Conservation. Conservation Biology 21: 956–963.

Magana M., Alonso J.C., Martín C.A., Bautista L.M. & Martín B. 2010. Nest-site selection by Great Bustards *Otis tarda* suggests a trade-off between concealment and visibility. Ibis 152:77-89.

Margalid, A. & Bertran J. 2000. Nest-building behaviour of the Bearded Vulture *Gypaetus barbatus*. Ardea 88: 259-264.

Margalida A., González L.M., Sánchez R., Oria J., Prada L., Caldera J., Aranda A. & Molina J.I. 2007. A long-term scale study of the breeding biology of Spanish imperial eagle *Aquila adalberti*. Journal of Ornithology 148: 309-322.

Margalida A., Heredia R., Razin M. & Hernández M. 2008. Sources of variation in mortality of the Bearded Vulture *Gypaetus barbatus* in Europe. Bird Conservation International 18: 1–10.

Margalida A., Donázar J. A., Bustamante J., Hernández F. & Romero-Pujante M. 2008. Application of a predictive model to detect long-term changes in nest-site selection in the Bearded Vulture *Gypaetus barbatus*: conservation in relation to territory shrinkage. Ibis 150: 242-249.

Margalida A., Moreno-Opo R., Arroyo B.E. & Arredondo A. 2011. Reconciling the conservation of an endangered species with an economically important anthropogenic activity: interactions between cork exploitation and the Cinereous Vulture *Aegypius monachus* in Spain. Animal Conservation 14: 167-174.

Martínez-Abraín A., Oro D., Jiménez J., Stewart G. & Pullin A. 2010. A systematic review of the effects of recreational activities on nesting birds of prey. Basic and Applied Ecology 11:312-319.

Ministry of Environment, Rural and Marine Affairs. 2011. Royal Decree 139/2011, 4th February, developing the List of Special Protected Wild Species and the Spanish Catalogue of Endangered Species. Boletín Oficial del Estado 46:20912-20951.

Morán-López R., Sánchez J.M., Costillo E. & Villegas A. 2006a. Nest-site selection of endangered cinereous vulture *Aegypius monachus* populations affected by anthropogenic disturbance: present and future conservation implications. Animal Conservation 9: 29-37.

Morán-López R., Sánchez J.M., Costillo E., Corbacho C. & Villegas A. 2006b. Spatial variation in anthropic and natural factors regulating the breeding success of the cinereous vulture *Aegypius monachus* in SW Iberian Peninsula. Biological Conservation 130:169-182.

Moreno-Opo R. & Guil F. (eds) 2007. Handbook on habitat and populations management of cinereous vulture in Spain, Ministry of Environment, Madrid.

Moreno-Opo R., Fernández-Olalla M., Margalida A., Arredondo A. & Guil F. 2012. Effect of methodological and ecological approaches on heterogeneity of nest-site selection of a long-lived vulture. PLoS ONE 7: e33469.

Newton I. 1979. Population Ecology of Raptors, T. & A.D. Poyser, London.

Oro D., Margalida A., Carrete M., Heredia R. & Donázar J.A. 2008. Testing the goodness of supplementary feeding to enhance population viability in an endangered vulture. PLoS ONE 3:e4084.

Ortega E., Mañosa S., Margalida A., Sánchez R., Oria J. & González L.M. 2009. A demographic description of the recovery of the Vulnerable Spanish imperial eagle *Aquila adalberti*. Oryx 43:113-121.

Pérez M.C. 2007 Diversity in the cork oak forests of the northern Straits of Gibraltar Region Southern Spain. Plant Ecology 189:87-100.

Poizaridis K., Goutner V., Skartsi T. & Stamou G. 2004. Modelling nesting habitat as a conservation tool for the Eurasian black vulture *Aegypius monachus* in Dadia Nature Reserve, northeastern Greece. Biological Conservation 118:235-248.

Pullin A.S., Knight T.M., Stone D.A. & Charman K. 2004. Do conservation managers use scientific evidence to support their decision-making? Biological Conservation 119:245–252.

R Development Core Team. 2008. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org.

Ricklets R.E. & Wikelski M. 2002. The physiology/lifehistory nexus. Trends in Ecology and Evolution 17:462-468.

Robinson S., Chiaradia A. & Hindell M.A. 2005. The effect of body condition on the timing and success of breeding little penguins *Eudryptula minor*. Ibis 147:483-489.

Sanz-Aguilar A., Tavecchia G., Pradel R., Mínguez E. & Oro D. 2008. The cost of reproduction and experience-dependent vital rates in a small petrel. Ecology 89: 3195-3203.

Sutherland W.J. 2007 Future directions in disturbance research. Ibis 149:120-124.

Terrasse M., Sarrazin F., Choisy J.P., Clemente C., Henriquet S., Lecuyer P., Pinna J. L. & Tessier C. 2004. A success story: the reintroduction of Eurasian griffon *Gyps fulvus* and black *Aegypius monachus* vultures to France. In: Chancellor R.D. & Meyburg B.U. (eds). Raptors worldwide: proceedings of the VI World Conference on Birds of Prey and Owls, World Working Group on Birds of Prey and Owls and MME/Birdlife Blackwell, Berlin and Budapest, pp. 127-145..

Velando A. & Freire J. 2001. How general is the central-periphery distribution among seabird colonies? Nest-spatial pattern in the European Shag. Condor 103:544-554.

Venables W.N. & Ripley B.D. 2002. Modern Applied Statistics with R. Fourth Edition, Springer, New York.

Wilkin T.A., King L.E. & Sheldon B.C. 2009. Habitat quality, nestling diet, and provisioning behaviour in great tits *Parus major*. Journal of Avian Biology 40:135-145.

Zuberogoitia I., Zabala J., Martinez J.A., Martinez J.E. & Azkona A. 2008. Effect of human activities on Egyptian vulture breeding success. Animal Conservation 11:313-320.

CAPÍTULO-CHAPTER
3

Foraging range and diet of cinereous vulture using livestock resources in central Spain

by

Rubén Moreno-Opo, Ángel Arredondo & Francisco Guil (2010)

Ardeola **57: 111-119.**

Área de campeo y alimentación del buitre negro según recursos ganaderos en el centro de España

Se analizan las áreas de campeo del buitre negro en las que se alimenta de cadáveres de ganado doméstico, en una colonia del centro-sur de España, mediante un método indirecto de estima de las zonas de movimiento a través de localizaciones de alimentación. Entre 2004 y 2008 se recogieron 377 crotales en las inmediaciones de nidos y posaderos de la especie, y se averiguó su procedencia geográfica. Los códigos individualizados de los crotales permitieron obtener localizaciones donde los buitres negros se alimentaron, a escala de explotación ganadera. La distancia media entre los lugares de recogida y de procedencia del crotal fue de 26,3 km (SD = 36,1). Los vuelos lineales oscilaron entre los 7,9 km y los 342 km. El polígono mínimo convexo de todas las localizaciones resultó de 66.732,28 km2. El área de campeo en que los buitres negros se alimentaron de carroñas de ganado (al 95 % de probabilidades, según el análisis Kernel) alcanzó 152.290,13 ha. Se discute la metodología empleada en este trabajo comparándolo con otros estudios similares que usaron el radioseguimiento como técnica de localización. Las diferencias de método han supuesto probablemente que las áreas de campeo y las distancias de vuelo obtenidas resultaran distintas. Estas consecuencias se analizan en relación a los tipos de presa disponibles, a los aprovechamientos económicos del medio, al tipo de hábitat y a la presencia de fuentes predecibles de alimento.

Ardeola 57(1), 2010, 111-119

FORAGING RANGE AND DIET OF CINEREOUS VULTURE *AEGYPIUS MONACHUS* USING LIVESTOCK RESOURCES IN CENTRAL SPAIN

ÁREA DE CAMPEO Y ALIMENTACIÓN DEL BUITRE NEGRO *AEGYPIUS MONACHUS* SEGÚN RECURSOS GANADEROS EN EL CENTRO DE ESPAÑA

Rubén MORENO-OPO* [1], Ángel ARREDONDO* and Francisco GUIL*

SUMMARY.—*Foraging range and diet of cinereous vulture* Aegypius monachus *using livestock resources in central Spain.*

We analysed the foraging area of cinereous vultures from a breeding colony of central Spain which were feeding on livestock carcasses by an indirect method that estimated movement ranges and feeding locations. Between 2004 and 2008 we checked the origin of 377 cattle tags, collected at nests and perches of the species. Using the individual codes of tags, we obtained the location (livestock exploitation) where the vultures fed. The average distance of the locations was 26.3 km (SD = 36.1). The linear flight routes ranged from 7.9 km to 342 km from the point of tag collection. The minimum convex polygon of all locations was 66,732.28 km^2. The home range of cinereous vultures feeding on carcasses (95 %, analysis Kernel) was 152,290.13 ha. We discuss the methodology used in this study compared to others using radiotracking as the location technique. Differences between methods probably explain why estimates of foraging areas and distances varied widely. These differences are analysed in relation to the different prey categories, the economic land use, type of habitat and the presence of predictable sources of food.

Key words: *Aegypius monachus*, carcass, cattle tags, cinereous vulture, diet, livestock, foraging range.

RESUMEN.—*Área de campeo y alimentación del buitre negro* Aegypius monachus *según recursos ganaderos en el centro de España.*

Se analizan las áreas de campeo del buitre negro en las que se alimenta de cadáveres de ganado doméstico, en una colonia del centro-sur de España, mediante un método indirecto de estima de las zonas de movimiento a través de localizaciones de alimentación. Entre 2004 y 2008 se recogieron 377 crotales en las inmediaciones de nidos y posaderos de la especie, y se averiguó su procedencia geográfica. Los códigos individualizados de los crotales permitieron obtener localizaciones donde los buitres negros se alimentaron, a escala de explotación ganadera. La distancia media entre los lugares de recogida y de procedencia del crotal fue de 26,3 km (SD = 36,1). Los vuelos lineales oscilaron entre los 7,9 km y los 342 km. El polígono mínimo convexo de todas las localizaciones resultó de 66.732,28 km^2. El área de campeo en que los buitres negros se alimentaron de carroñas de ganado (al 95 % de probabilidades, según el análisis Kernel) alcanzó 152.290,13 ha. Se discute la metodología empleada en este trabajo

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comparándolo con otros estudios similares que usaron el radioseguimiento como técnica de localización. Las diferencias de método han supuesto probablemente que las áreas de campeo y las distancias de vuelo obtenidas resultaran distintas. Estas consecuencias se analizan en relación a los tipos de presa disponibles, a los aprovechamientos económicos del medio, al tipo de hábitat y a la presencia de fuentes predecibles de alimento.

Palabras clave: *Aegypius monachus*, área de campeo, buitre negro, carroña, crotal, dieta, ganado.

INTRODUCTION

Prey abundance, distribution, ocurrence and size are likely to influence foraging areas and home range characteristics of scavenging raptor species (Costillo *et al.,* 2007; Margalida *et al*., 2009) and, therefore, determine their feeding behaviour and associated territoriality (Hiraldo, 1977; Donázar, 1993). Foraging areas of the cinereous vulture have been studied in different regions where the species occurs (Carrete and Donázar, 2005; Costillo, 2005; Vasilakis *et al*., 2006). Efforts have been mainly directed to knowing the extension of these areas and to evaluate the impact that food availability has on them, as well as variations depending on the season and invidual age classes. This knowledge is the basis for assessing food resource availability, habitat selection patterns and the influence of different threats.

We report the locations where cinereous vultures fed on livestock carcasses in relation to their breeding colony in south-central Spain. The aim is to define the extension and location of feeding areas of the species in order to implement efficient management and conservation strategies.

MATERIAL AND METHODS

Fieldwork was carried out in the Umbría de Alcudia cinereous vulture breeding colony, in Ciudad Real province (figure 1). This colony holds the fourth largest aggregation in Spain, with 129 pairs in 2006 (De la Puente *et al*., 2007). This is an area mainly covered by Mediterranean forest, with tree species including *Quercus suber*, *Quercus rotundifolia*, *Junniperus communis* and *Arbutus unedo*, and a thick bush and understory development. Nests are located on mountain slopes ranging in altitude between 736 and 1.178 m a.s.l. This area is inhabited by significant populations of wild ungulates (i.e., red deer *Cervus elaphus* and wild boar *Sus scropha*) and lagomorphs, while in the surrounding areas there are *dehesas* –parkland with varying densities of trees and agricultural land with abundant livestock populations. Between 2004 and 2008, in the period October-December, cinereous vulture food remains were collected from close to 88 nests and 11 perches in the colony. 480 cattle tags were found among the food remains. The origin of animals that wear cattle tags can be identified, because these plastic marks have the individual codes of the farm of origin and the municipality in which it is located. The cinereous vultures ingest these tags, which are later regurgitated in pellets. Therefore, cattle tags show at a livestock farm scale the location where cinereous vultures have ingested the prey.

In order to know the exact origin of the cattle tags an information request was addressed to the Official Livestock Administrations. This search revealed the origin of 377 units (78.5 % of the total). The species that carries the plastic tag is known based on the use described in the official databases of the farms and/or through the written code on the tag.

Cinereous vulture individuals that ingested cattle tags were any individuals that had been

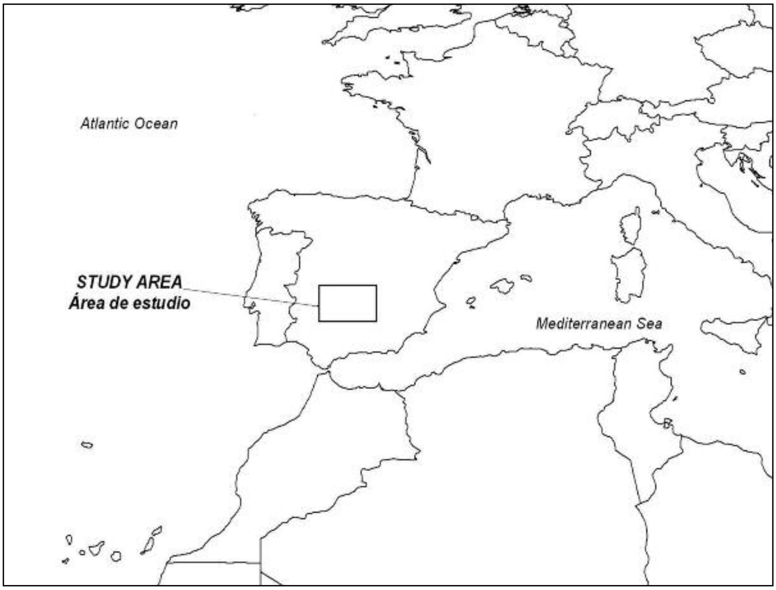

FIG. 1.—Study area in the Iberian Peninsula, including the cinereous vulture *Aegypius monachus* breeding colony and the foraging locations obtained through the collection of livestock tags.
[Área de estudio del presente trabajo, incluyendo la colonia de buitre negro Aegypius monachus *objeto de seguimiento y las localizaciones de presencia obtenidas.]*

presented in the breeding colony during the study period. Results do not show the foraging areas of specific age classes or sexes. However, given that 95 % of the cattle tags had been collected directly under nest sites, these suggesting that individuals that had ingested the tags were probably established as breeders in the colony.

The feeding area has been considered as the total surface where feeding locations have been registered. It has been expressed through the Minimum Convex Polygon (MCP; Mohr, 1947). MCP is defined as the area formed by the outer polygon encompassing all the points where the studied animals have been located. The foraging range is referred to as the probability of finding locations using Kernel polygons (Worton, 1989; White and Garrot, 1990). These are surfaces delimited by probability isolines of finding the locations of the studied animals; in most studies it is shown the 95 % of probability though the present work also offers other probability values. The spatial analysis has been conducted with ArcView GIS 3.1. software, Animal Movement extension. Similarly, distances travelled by the birds were obtained, taking as point of origin the location where cattle tags had been collected in the colony and as destination the geographic centre of the farm of origin of the tag.

RESULTS

Cinereous vultures of the Umbría de Alcudia colony feed in an area of 66,732.28 km^2 (table 1). There have been feeding points in the provinces of Ciudad Real, Córdoba, Badajoz, Jaén, Cuenca and Cáceres. The foraging range of the studied individuals, in a 95 % probability Kernel polygon (White y Garrot 1990), has resulted 152.290,13 ha (table 1, figure 2).

TABLE 1

Foraging and feeding areas and distances of the foraging locations of cinereous vultures *Aegypius monachus* in central Spain.
[Áreas de alimentación y distancias entre las localizaciones de alimentación de buitres negros Aegypius monachus *en España central.]*

Feeding area (MPC, km^2) *Área de alimentación (MPC, km^2)*		66,732.28
Foraging range (ha) *Área de campeo (ha)*	95 %	152,290.13
	90 %	83,516.97
	80 %	51,172.54
	70 %	40,640.08
	60 %	32,463.15
	50 %	25,401.11
Average distance among foraging and tags collection locations (km) *Distancia promedio de localizaciones de alimentación al punto de recogida de crotales (km)*		26.3 (± 36,1)
Maximum foraging distance (km) *Distancia máxima de alimentación (km)*		342
Minimum foraging distance (km) *Distancia mínima de alimentación(km)*		7.9

It was possible to determine the species for 74.8 % of the tags. 54.6 % of the tags were put in pigs, 37.1 % in sheep/goats and 8.3 % in cattle.

DISCUSSION

The method assigning feeding locations in this study is novel and different with respect to the majority of previous works (Corbacho *et al.,* 2004; Carrete and Donázar, 2005; Costillo, 2005; Vasilakis *et al*., 2006) which determined foraging areas based on the telemetry of specific individuals. Radiotracking allows a larger number of locations of the different studied cinereous vultures -1.219 (Corbacho *et al*., 2004), at least 673 (Carrete and Donázar, 2005) and 929 (Vasilakis *et al*., 2006) vs. 377 cattle tags recovered over five years of work in the present study. Moreover, and taking into account the errors assigned to the technique (White and Garrot, 1990; Kenward, 2001), telemetry enables precise location of the foraging areas of the studied species. Furthermore, radiotracking specific individuals allows assessment of possible variations in foraging ranges among age classes and sexes and at different times of the year (Corbacho *et al.,* 2001; Costillo *et al*., 2007), while the impos-

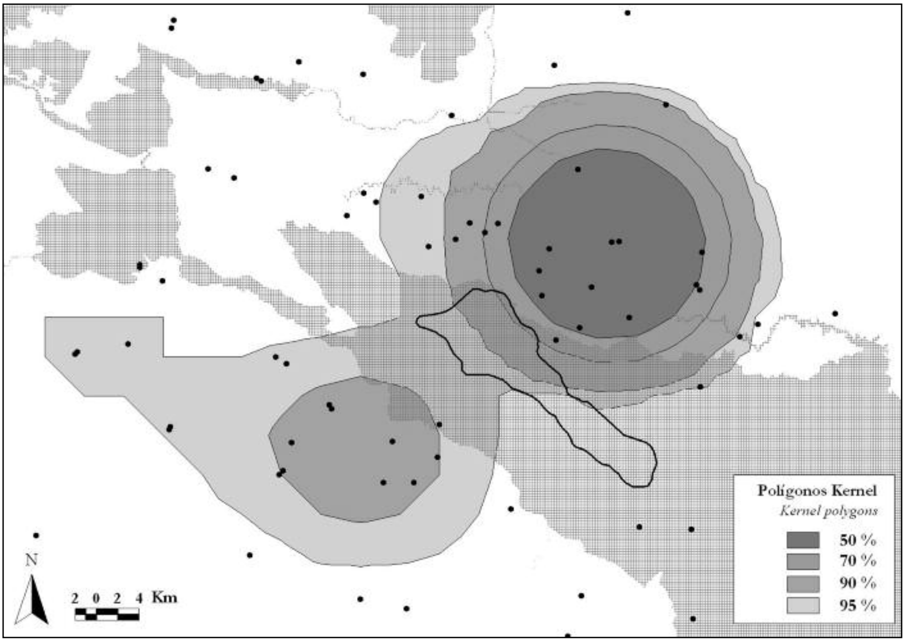

FIG. 2.—Home range of cinereous vulture *Aegypius monachus* in the studied breeding colony. It includes the different probabilities of occurrence in Kernel polygons (ArcView GIS 3.0, Animal Movement), the outline of the studied colony (thick black line), the origin of the cattle tags (black dots) and the Natura 2000 network in the area (grey shading).
[Áreas de campeo de los buitres negros Aegypius monachus *presentes en la colonia objeto de estudio, expresado según las distintas probabilidades de aparición en polígonos Kernel (ArcView GIS 3.0, Animal Movement). Se muestra el contorno de la colonia estudiada (línea negra gruesa), las localizaciones de procedencia de los crotales (puntos negros) y la trama de espacios incluidos en Natura 2000.]*

sibility of knowing which individual has ingested the cattle tag prevents carrying out these comparisons. Finally, the main limiting factor for using cattle tags in these studies is that it is not possible to track the searching movements for finding non ear-tagged dead animals, mainly wild ungulates and lagomorphs, which are of great importance in the diet of the cinereous vulture (Corbacho *et al*., 2007). This limits the general conclusions about the entire foraging area used by the species. Nevertheless, the use of cattle tags as location technique has several methodological advantages. The samples correspond to the foraging pattern of a large number of individuals, since they were collected in 88 different nests and 11 perches over five consecutive years of work. In the previously cited works, the maximum number of tracked cinereous vultures was 14 (Carrete and Donázar, 2005). In addition, to know the origin of the tags involves locating an unmistakable feeding point; scavengers ingest these tags along with the whole biomass.

The foraging area of the cinereous vulture estimated herewith is larger compared with that obtained in other studies (table 2). The size of the foraging range could be related to the number of studied individuals, as well as to the relatively large population size of the studied colony, the period of the annual cycle or the availability of food resources (Costillo, 2005; Costillo *et al.*, 2007). In this work, the number of vultures carrying cattle tags to the collection points is unknown. Therefore, results are expressed as the total of all the individuals of the breeding colony. For this reason, as there is greater potential variability in the foraging behaviour, the foraging range should have been larger than that obtained by tracking a few individuals (Carrete and Donázar, 2005; Vasilakis *et al.*, 2006). When pooling breeding and non-breeding birds, the size of the feeding area increases (Corbacho *et al.,* 2004), giving similar values to those obtained in the present work. However, the reported areas could have been overestimated by not registering the feeding on potential wild prey species abundant in the close vicinity of the colony. The Kernel probability polygons are determined by the number of locations, so that when no records exist in the close surroundings of the colony, these polygons could cover wider and remote areas based exclusively in the presence of livestock. The nearest areas around the breeding nuclei offer only wild prey to cinereous vultures – rabbit *Oryctolagus cuniculus*, Spanish ibex *Capra hispanica*, red deer and wild boar. These areas are covered by mature Mediterranean forests and rangelands with no livestock exploitation. Therefore, these two factors (type of available prey and habitat) are of great importance for the configuration of the foraging areas because they determine search patterns (Carrete and Donázar, 2005) and could reflect an increase in their extension (table 2).

The distances travelled while foraging are on average higher than those of other similar studies (26,3 ± 36,1 –SD– km, as compared to 16,1 ± 14,0 –SD– km in Corbacho *et al.*, 2004).

It has to be noted that 54.5 % of the obtained locations came from a single vulture restaurant, with a continuous input of food; this illustrating the importance of these places for the management and conservation of scavenging birds (Donázar, 1992; Camiña and Mon-

TABLE 2

Home range (ha) of cinereous vulture *Aegypius monachus* individuals, in 95 % Kernel polygon (Arc View GIS 3.1). References: [1] Costillo, 2005; [2] Carrete y Donázar 2005; [3] Vasilakis *et al.* 2006; [4] present study.

[Área de campeo (en ha.) de individuos de buitre negro Aegypius monachus*, considerando el polígono Kernel al 95 % (Arc View GIS 3.1). Fuentes: [1] Costillo, 2005; [2] Carrete y Donázar 2005; [3] Vasilakis* et al. *2006; [4] presente estudio.]*

Study area *Área*	**Breeding Seaton** *Época reproductora*	n	**Non breeding Seaton** *Época no reproductora*	n
Sierra de San Pedro[1]	66,755 ± 71,397	6	15,526 ± 4,240	4
Sierra Pelada[2]	135,430 ± 58,965	14	77,775 ± 35,021	6
Dadia National Park[3]	61,203 ± 22,391	6		
Umbría de Alcudia[4]		152,290.13		

telío, 2006; Carrete *et al*., 2006; Moreno-Opo *et al*., 2007). This has probably altered the natural foraging and movement patterns, these being routinely directed towards a specific point where food is abundant and predictable (Robb *et al*., 2008; Carrete *et al.,* 2009). On the other hand, it would be interesting to evaluate the relationship between livestock availability in the foraging areas and the diet proportion obtained in the present work (Savage, 1931) in order to check the selection of some prey categories or to test the influence of the official management procedures towards different kind of carcasses (García de Francisco and Moreno-Opo, 2009; Moreno-Opo *et al.* 2010).

Cinereous vulture conservation depends on the land management where it is to be found. While breeding sites have a strictly enforced protection level (Moreno-Opo, 2007), management efforts in foraging areas are still incipient (Carrete and Donázar, 2005). The main threats to this and other scavenging raptor species come from the scarcity and poor quality of the food, including poisoning (González and Moreno-Opo, 2008; Hernández and Margalida, 2008; Margalida *et al*., 2008, Hernández and Margalida, 2009). Therefore, one of the most effective conservation measures is evaluating food resources, ensuring their availability and quality (Donázar *et al*., 2009). The first step to undertake these conservation actions is to know and to delimit areas where the species is present at different life cycle periods.

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BIBLIOGRAPHY

CAMIÑA, A. and MONTELÍO, E. 2006. Griffon vulture *Gyps fulvus* food shortages in the Ebro Valley (NE Spain) caused by regulations against Bovine Spongiform Encephalopathy (BSE). *Acta Ornithologica,* 41: 1-8.

CARRETE, M. and DONÁZAR, J. A. 2005. Application of central-place foraging theory shows the importance of Mediterranean dehesas for the conservation of cinereous vulture *Aegypius monachus*. *Biological Conservation,* 126: 582-590.

CARRETE, M., DONÁZAR, J. A. and MARGALIDA, A. 2006. Density-dependent productivity depresión in Pyrenean bearded vultures: implications for conservation. *Ecological Applications,* 16: 1674-1682.

CARRETE, M., MARGALIDA, A., ORO, D. and DONÁZAR, J. A. 2009. Population consequences of vulture restaurants: the bearded vulture *Gypaetus barbatus* in the Spanish Pyrenees. In, J. A. Donázar, A. Margalida and D. Campión (Eds.). *Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology.* Munibe, 29. Sociedad de Ciencias Aranzadi. Donostia.

CORBACHO, C., COSTILLO, E., LAGOA, G. and MORÁN, R. 2001. Effect of Breeding Cycle on foraging areas and home-range of black vulture *Aegypius monachus* in Extremadura. In, *Proceedings of 4th Eurasian Congress on Raptors*. EBD-Raptor Research Foundation. Seville.

CORBACHO, C., COSTILLO, E. and MORÁN, R. 2004. Foraging areas and home-range of black vultures *Aegypius monachus*: a preliminary analysis. In: *Proceedings of International Symposium on the Black Vulture* Aegypius monachus. Córdoba.

CORBACHO, C., COSTILLO, E. and PERALES, A. B. 2007. Alimentación del buitre negro en la península Ibérica. In, R. Moreno-Opo and F. Guil (Coords.) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

COSTILLO, E. 2005. *Biología y conservación de las poblaciones de buitre negro* Aegypius monachus *en Extremadura*. PhD Thesis. University of Extremadura. Badajoz.

COSTILLO, E., CORBACHO, C., SÁNCHEZ, J. M. and VILLEGAS, A. 2007. Áreas de campeo. In, R. Moreno-Opo and F. Guil (Coords.) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

DE LA PUENTE, J., MORENO-OPO, R. and DEL MORAL, J. C. 2007. *El buitre negro en España. Censo Nacional 2006*. SEO/BirdLife. Madrid.

DONÁZAR, J. A. 1992. Muladares y basureros en la biología y conservación de las aves en España. *Ardeola,* 39: 29-40.

DONÁZAR, J. A. 1993. *Los buitres ibéricos*. J. M. Reyero (Ed.). Madrid.

DONÁZAR, J. A., MARGALIDA, A. and CAMPIÓN, D. (Eds.) (2009). *Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology.* Munibe, 29. Sociedad de Ciencias Aranzadi. Donostia.

GARCÍA DE FRANCISCO, J. M. and MORENO-OPO, R. 2009. Livestock carcasses management today: is there enough flexibility to deal with new conservation strategies? In: Donázar, J. A., Margalida, A. and Campión, D. (Eds.). *Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology.* Munibe, 29. Sociedad de Ciencias Aranzadi. Donostia.

GONZÁLEZ, L. M. and MORENO-OPO, R. 2008. Impacto de la falta de alimento en las aves necrófagas amenazadas. *Ambienta,* 73: 48-55.

HERNÁNDEZ, M. and MARGALIDA, A. 2008. Pesticide abuse in Europe: effects on the cinereous vulture *Aegypius monachus* in Spain. *Ecotoxicology,* 17: 264-272.

HERNÁNDEZ, M. and MARGALIDA, A. 2009. Poison related-mortality effects in the endangered Egyptian vulture *Neophron percnopterus* in Spain. *European Journal of Wildlife Research,* 55: 113-121.

HIRALDO, F. 1977. *El buitre negro* Aegypius monachus *en la península Ibérica*. PhD Thesis. University of Seville. Seville.

KENWARD, R. E. 2001. *A manual for wildlife radiotracking*. Academic Press. San Diego.

MARGALIDA, A., HEREDIA, R., RAZIN, M. and HERNÁNDEZ, M. 2008. Sources of variation in mortality of the Bearded Vulture *Gypaetus barbatus* in Europe. *Bird Conservation International*, 18: 1-10.

MARGALIDA, A., BERTRÁN, J. and HEREDIA, R. 2009. Diet and food preferences of the endangered Bearded Vulture *Gypaetus barbatus*: a basis for their conservation. *Ibis*, 151: 325-243.

MOHR, C. O. 1947. Table or equivalent populations of north american small mammals. *American Naturalist*, 37: 223-249.

MORENO-OPO, R. 2007. El buitre negro. In, R. Moreno-Opo and F. Guil (Coords.). *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

MORENO-OPO, R., SAN MIGUEL, A. and CAMIÑA, A. 2007. Ganadería y buitre negro. In, R. Moreno-Opo y F. Guil (Coords.) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

MORENO-OPO, R., MARGALIDA, A., ARREDONDO, A., GUIL, F., MARTÍN, M., HIGUERO, R., SORIA, C. and GUZMÁN, J. 2010. Factors influencing the presence of the cinereous vulture *Aegypius monachus* at carcasses: food preferences and implications for the management of supplementary feeding sites. *Wildlife Biology*, 16: 25-34.

SAVAGE, R. E. 1931. The relation between the feeding of the herring off the east coast of England and the plankton of the surrounding waters. *Fishery Investigation, Ministry of Agriculture, Food and Fisheries*, *Series 2* 12: 1-88.

ROBB, G. N., MCDONALD, R. A., CHAMBERLAIN, D. E. and BEARHOP, S. 2008. Food for thought: supplementary feeding as a driver of ecological change in avian populations. *Frontiers in Ecology Environment*, 6: 476-484.

VASILAKIS, D., POIZARIDIS, K. and ELORRIAGA, J. 2006. Breeding season range use of a Eurasian Black Vulture *Aegypius monachus* population in Dadia National Park and the adjacent areas, Thrace, NE Greece. In, D. C. Houston and S. E. Piper (Eds.). *Proceedings of the International Conference on Conservation and Management of Vulture Populations*. Natural History Museum of Crete. WWF Greece. Thessaloniki.

WHITE, G. C. and GARROT, R. A. 1990. *Analysis of Wildlife Radio-Tracking data*. Academic Press. New York.

WORTON, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. *Ecology,* 70: 164-168.

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CAPÍTULO-CHAPTER
4

Factors influencing the presence of cinereous vulture at carcasses: food preferences and implications for the management of supplementary feeding sites

by

Rubén Moreno-Opo, Antoni Margalida, Ángel Arredondo, Francisco Guil, Manuel Martín, Rafael Higuero, Carlos Soria & José Guzmán (2010)

Wildlife Biology **16: 25-34**

Factores que influyen en la presencia de buitres negros en carroñas: preferencias alimentarias e implicaciones para la gestión de puntos de alimentación suplementaria

Se estudiaron los factores que determinan la presencia del buitre negro en 134 aportes de carroña depositados experimentalmente en distintas Zonas de Especial Protección para las Aves (ZEPA) del oeste y centro de España. Nuestros objetivos fueron evaluar el aprovechamiento de estos cadáveres y otros subproductos animales con el fin de averiguar las preferencias de alimentación del buitre negro y, de este modo, formular recomendaciones para la gestión de puntos de alimentación suplementaria específicos para esta especie de buitre. Los resultados sugieren que el número de buitres negros que acuden a alimentarse en los aportes de carroña está relacionado con la cantidad de biomasa existente y con los tipos de trozos y partes en que se presenta el alimento. El buitre negro selecciona piezas musculares de tamaño mediano y pequeños trozos periféricos de carne y tendón. El tiempo que transcurre antes de que los buitres negros comiencen a consumir la carroña depende de la cantidad de biomasa existente, del número de trozos en las que esta carroña esté dividida, y del tipo de formato en que se suministre (trozos pequeños y medianos más favorables que piezas grandes; y trozos más dispersos más favorable que todos en un mismo punto). Por otro lado, la densidad poblacional relativa de la especie en nuestra área de estudio y la fase del ciclo reproductivo en que se encuentre parecen determinar el tiempo invertido en la alimentación sobre los cadáveres. Estos resultados pueden ayudar a los gestores a optimizar la creación y el manejo de puntos de alimentación suplementaria para favorecer su uso por el buitre negro.

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Factors influencing the presence of the cinereous vulture *Aegypius monachus* at carcasses: food preferences and implications for the management of supplementary feeding sites

Rubén Moreno-Opo, Antoni Margalida, Ángel Arredondo, Francisco Guil, Manuel Martín, Rafael Higuero, Carlos Soria & José Guzmán

We studied the factors that determine the presence of the cinereous vulture *Aegypius monachus* at 134 carcasses experimentally distributed in Special Protection Areas for Birds (SPA) in western and central Spain. Our goals were to assess the use of these carcasses and by-products in order to find out the cinereous vulture's food preferences and thus provide recommendations for the management of specific vulture restaurants for this species. Our results suggest that the number of cinereous vultures that come to feed on the carcasses is related to the quantity of biomass present and to the types of pieces of the provided food. Cinereous vultures prefer individual, medium-sized muscular pieces and small peripheral scraps of meat and tendon. The time that elapses before the cinereous vultures begin to consume a carcass depends on the biomass delivered, the number of pieces into which it is divided, and the type categories of the provided food. The population density of the species in our study area and the breeding stage seem to determine the time invested in feeding at the carcasses. These results may help managers to optimise the creation of vulture restaurants and favour their use by cinereous vultures.

Key words: Aegypius monachus, carrion, cinereous vulture, feeding sites, food preferences

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In Europe, vultures have mainly fed on carrion from wild and domestic ungulates. The main sources of the carcasses were the death of wild ungulates (natural and non-natural mortality through hunting practices) and extensive livestock grazing (Donázar et al. 2009a). Alternative food sources included the traditional 'muladar' (place where livestock carcasses were traditionally dumped so that scavenger birds could eat them and thus get rid of them) and, more recently, artificial supplementary feeding sites. The supplementary feeding sites were created in an effort to increase the vulture populations and their breeding parameters, as well as to facilitate the species' geographical expansion and reduce the risks of consumption of micro-pathogen/pesticide contaminated prey (Donázar et al. 2009a). However, since 2004, the appearance of Bovine Spongiform Encephalopathy (BSE) significantly reduced the presence of food in the field. The precautionary principle of banning the dumping of animals that died in the field in order to avoid zoonoses and livestock transmissible diseases spreading, and the decrease in trophic availability due to the closing of the 'muladares', may affect the feeding habits and foraging behaviour of scavengers and thus have major implications for the conservation of endangered vultures (Tella 2001, García de Francisco & Moreno-Opo 2009, Donázar et al. 2009b). In this respect, the sudden changes in the availability of food may cause changes in the species' population dynamics (Ostfeld & Keesing 2000). In the case of scavenger birds, a rapid significant reduction in food availability can have a negative impact on population parameters, since these are K-selected species characterised by long life cycles and low fecundity rates (Donázar 1993, Moreno 2002).

In Spain, the implementation of public health regulations has led to a reduction in the availability of livestock carcasses (Camiña & Montelío 2006, Moreno-Opo et al. 2007). This has resulted in an increase in the number of malnourished young vultures being taken to wildlife recovery centres, and an increase in the number of reports of attacks on neonatal and non-neonatal livestock by Eurasian griffon *Gyps fulvus* and cinereous *Aegypius monachus* vultures (see Donázar et al. 2009a).

The management of trophic resources is of great importance for the conservation of threatened species (BirdLife International 2004, Jones 2004) such as the cinereous vulture, a species considered 'Near Threatened' by IUCN (BirdLife International 2009). The Spanish cinereous vulture population is estimated at 1,845 pairs, which represents 98% of the European population and between 18-25% of the world population (De la Puente et al. 2007, Moreno-Opo 2007). Some of the main threats come from the lack of natural food and its poor quality (Sánchez 2004) as well as poisoning from the consumption of pesticide-contaminated prey (Hernández & Margalida 2008). Thus, the application of measures for the management of trophic resources through feeding stations may constitute an effective conservation tool.

The cinereous vulture mainly feeds on the carcasses of rabbits, sheep and wild ungulates (see Hiraldo 1976, Corbacho et al. 2007). However, changes in the availability of prey over the last 30 years have led to a decrease in the number of rabbits in its diet and an increase in the consumption of domestic ungulates (Corbacho et al. 2007, Costillo et al. 2007). For the conservation of this species, detailed knowledge of its diet and which specific anatomic parts of a carcass it prefers may constitute a fundamental tool for the design of conservation strategies (see for example Margalida et al. 2009).

In our study, we aim to assess the use of carcasses and food preferences by cinereous vultures, and to provide recommendations for the future establishment of vulture restaurants. We obtained information directly in the field through the experimental placing of carcasses and by-products. Our results allow us to provide recommendations regarding how to optimise the future management of specific supplementary feeding stations for the management and conservation of the cinereous vulture.

Material and methods

Study area

The fieldwork was carried out in six Special Protection Areas for Birds (SPA, Fig. 1) in the regions of Extremadura and Castilla-La Mancha (western and central Spain). These are areas with rolling hills and mountains with vegetation dominated by holm oak *Quercus ilex* and cork oak *Q. suber*, in which most of the species' nests are located. In our study, the carcasses were delivered at altitudes ranging between 336 and 788 m a.s.l., in the proximity of six cinereous vulture breeding colonies, which included the four largest colonies in Spain: Sierra de San Pedro (336 pairs), Monfragüe (287 pairs), Cabañeros (165 pairs) and Umbría de Alcudia (129 pairs) (De la Puente et al. 2007). The distance between the

Figure 1. Study area with the six Special Protection Areas for Birds in which the carrion was delivered.

feeding sites and the nests occupied by the species ranged between 0.97 and 39.1 km.

Fieldwork and variables studied

Our study was carried out between December 2003 and December 2006. We monitored 134 carcasses, spread out homogeneously over the different months and years. The remains were delivered to 67 different sites in 13 private estates. The feeding sites were chosen as randomly as possible and were in no case determined by the presence of fenced-in 'muladares'. The carcasses were monitored by the observers, using 20-60 x telescopes at distances of > 500 m, so that their consumption and the vultures' behaviour could be studied without disturbing the birds. The species delivered as carrion were all present in our study area. The following species were delivered: 2,945 carcasses of red deer *Cervus elaphus*, 113 of sheep *Ovis aries*, 178 of wild boar *Sus scrofa*, one of pig *Sus scrofa* var. dom., 21 of fallow deer *Dama dama*, 14 of mouflon *Ovis musimon*, one of cow *Bos taurus*, and two of red fox *Vulpes vulpes*. Each carcass delivered was monitored for a maximum of 48 hours after it was deposited (carcasses generally were eaten during this period), and until it was consumed. Five response parameters were considered in order to assess the presence of the cinereous vulture at the studied carcasses, in accordance with their characteristics: 1) the number of cinereous vultures that came to the carcass, 2) the ratio of cinereous vultures to griffon vultures, 3) the ratio of non-adult cinereous vultures (juveniles and subadults pooled) to the adults (> 5 years), 4) the time the vultures took to start eating, considering this to be the interval of time between the moment the carcass was delivered until the first cinereous vulture started to eat, and 5) how long the birds ate for, considering this to be the time that elapsed between the moment the first cinereous vulture started eating and the time when the last cinereous vulture at the carcass stopped eating. These parameters were related to a series of explanatory variables in order to analyse their influence on the presence of vultures (Table 1).

The number of nests around the site where the by-products were delivered was selected as an explanatory variable of the density of cinereous vultures, since, as a central foraging species, the nest is the origin of the foraging activity for territorial members of this species (Carrete & Donázar 2005). Based on the home ranges of individual cinereous vultures during the breeding season, which were obtained from the literature (Carrete & Donázar 2005, Costillo 2005, Vasilakis et al. 2006), we estimated the daily flight distance of the vultures at 16.4 km average radius (N = 3). Thus, this distance was considered a theoretical radius for calculating the number of nests present around the location of the delivered carcasses, in order to then divide the cinereous vulture population density into three categories (low, medium and high; see Table 1).

In order to discover how often the carcasses were used by different age classes (juvenile, subadult and adult) in accordance with breeding stage, observations were grouped into three periods: incubation (I: February-April), chick-rearing (CR: May-August), and the non-breeding period (NB: September-January), partially corresponding to post-fledging and pre-laying periods.

In parallel, in order to determine the most consumed parts of the supplied carcasses, a series of categories were established to approximate food preferences: 1) all kinds of remains from the carcass (vultures feeding indistinctly on all kinds of remains available, from the whole carcass to muscular pieces, entrails, etc. both concentrated in one place and/or scattered), 2) muscular pieces extracted from a whole carcass, 3) loose medium-sized muscular pieces (0.2-5 kg), 4) entrails in a whole carcass, 5) entrails scattered around a carcass, and 6) small peripheral scraps of meat and tendons. The observations of the different carcasses were independent and more than one category was noted, in accordance with the feeding activity displayed by the birds studied.

Statistical analyses

We conducted three different statistical analyses based on ecological issues considered in the study. First, in order to determine the appearance patterns of the cinereous vultures in accordance with the different explanatory variables considered, we fitted General Linear Models (GLM). This analysis aims to establish an applicable and predictable relationship between the presence of the cinereous vulture at carcasses (expressed as five response parameters) related to the different predictor explanatory variables (see Table 1). All the explanatory variables were included in the analysis as independents. The independent variables 'estate' and 'SPA' were nested as they were integrated, since the different estates were grouped together in each of the SPAs

Table 1. Independent variables assessed to analyse the presence of the cinereous vulture at carcasses through GLM analysis (* = continuous variable, ** = categorical variable), and description of the categories and the field of study referred to by the variables.

Variable	Categories	Description of the category or variable	Field of feeding ecology study
Format**	1	Whole carcass(es)	Carrion typology
	2	Whole carcass(es) and piled up scraps	
	3	Whole carcass(es) and scattered scraps (radius of up to 50 m)	
	4	Piled up scraps	
	5	Scattered scraps (radius of up to 50 m)	
Biomass (kg)*		Weight (kg) of the carcass delivered	
Number of items*		Number of different items into which the carcass delivered was divided	
Breeding period**	Non-breeding	Outside of the breeding period, post-fledging and pre-laying periods (September-January)	Time
	Incubation	Incubation period (February-April)	
	Chick-rearing	Period when chicks are reared in the nest (May-August)	
Time**		Time when the animal by-products were delivered	
Plant cover*		Percentage (%) of land covered by vegetation > 50 cm high in a 100 m radius around the centre of the carcass	Habitat
Cinereous vulture density **	Low	< 25 cinereous vulture nests within a 16.4 km radius around the feeding station	Population density
	Medium	26-75 cinereous vulture nests within a 16.4 km radius around the feeding station	
	High	> 75 cinereous vulture nests within a 16.4 km radius around the feeding station	
Estate**		Private estate in which the carcass was delivered	Location
SPA**		Special Protected Area (SPA) in which the carcass was delivered	

studied. The dependent variables were fitted to a binomial error, and a log link function was used. The normality of residuals of the response parameters analysed was studied, in order to check the required hypothesis to fit a GLM analysis. Five multi-relation parameter-independent response variable analyses were arranged to identify statistical significance of the variables. We first analysed the number of cinereous vultures attending to the carcasses in relation to the considered variables. The initial model included: 'format of the carcass', 'biomass delivered', 'number of items', 'plant cover', 'population density', 'breeding period', 'time' and 'SPA*estate' (nested). Then, we analysed four other response parameters: the ratio of cinereous vultures to griffon vultures at carcasses, the ratio of non-adult to adult cinereous vultures at carcasses, the time to arrival of cinereous vultures to begin feeding and the time spent by cinereous vultures feeding on the carcass. The initial model was the same as that described for the number of cinereous vultures attending to the carcasses. For the sake of clarity, in our presentation, we reported only the significant terms (factors and their interactions). All other factors not reported were non-significant ($P > 0.05$).

The second analysis was conducted to determine the differences in the percentage of visits by different age classes of cinereous vultures over the three phenological periods considered (non-breeding, incubation and chick-rearing). This was tested using analysis of variance (ANOVA). When the ANOVA results proved to be statistically significant, a post-hoc analysis was made employing the Scheffé test to identify differences between groups.

Finally, frequencies obtained in the observations of the type of categories fed on by the cinereous vultures were analysed using the χ^2 test. Values are presented as means ± SD.

Results

Of the 134 carcasses monitored, a total of 3,136 visits of cinereous vultures and 10,610 visits of

Table 2. Statistically significant relationships between the explanatory variables studied in relation to the response parameters, resulting from the GLM analysis.

Response parameter	Explanatory variable	Sum of squares	df	F	P
Cinereous vulture (N) that came to the carcass	Biomass delivered (kg)	55245.40	1 (114)	103.49	0.00001
	Format of carcass	6468.58	4 (114)	3.03	0.0215
Time (minutes) elapsed until starting eating	Biomass delivered (kg)	165054.30	1 (75)	7.22	0.0097
	Format of carcass	230198.16	4 (75)	2.52	0.0520
	Number of items of the carcass	238988.11	1 (75)	10.45	0.0022
Time (minutes) that cinereous vulture spent eating	Cinereous vulture density	87514.39	2 (74)	4.51	0.0158
	Breeding period	60211.66	2 (74)	3.11	0.0535
	Special Protected Area for birds	190670.06	7 (74)	2.81	0.0151

griffon vultures were recorded. The average number of cinereous vultures observed at each carcass was 23.40 ± 52.25 individuals (range: 0-400, N = 134), with the average maximum number of cinereous vultures observed simultaneously at a carcass being 21 ± 26 (range: 2-110, N = 87). The average time that elapsed between the carcass being delivered and it being eaten was 236.64 ± 134.42 minutes (range: 10-780, N = 79). The average time the birds spent at the carcass was 165.18 ± 113.68 minutes (range: 12-606, N = 78). The ratio of cinereous vultures vs griffon vultures present at the carcass was 1.29 ± 0.75 (range: 0-6.19, N = 95).

The results of the GLM analysis were significant for three of the five response parameters analysed in relation to the total number of studied variables: the number of cinereous vultures that visited the carcasses ($F = 23.05$, df = 24, 114, $P = 0.00001$, adjusted $R^2 = 82.27\%$), the time that elapsed between the delivery of the carcass until the first cinereous vulture started eating ($F = 1.83$, df = 24, 75, $P = 0.035$, adjusted $R^2 = 21.06\%$) and the time invested by the cinereous vultures in eating each carcass ($F = 2.17$, df = 24, 74, $P = 0.010$, adjusted $R^2 = 27.47\%$).

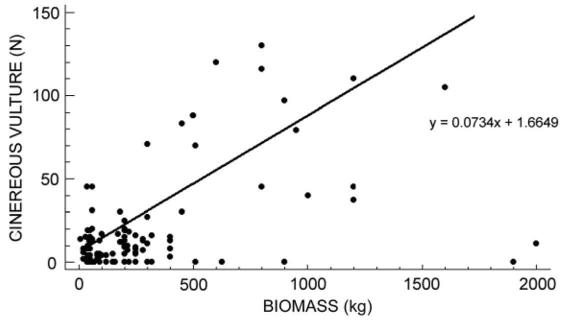

Figure 2. Relationship between the total number of cinereous vultures and the biomass present (kg) at the carcasses studied.

The number of cinereous vultures that visited the carcasses was positively related to the biomass delivered, and significantly related to the format of the carcass (Table 2, Figs. 2 and 3a). The time that elapsed between the delivery of the carcass until the first cinereous vulture started eating was significantly related to the biomass delivered and the number of items (see Table 2), and marginally related with the format (see Table 2 and Fig. 3). The

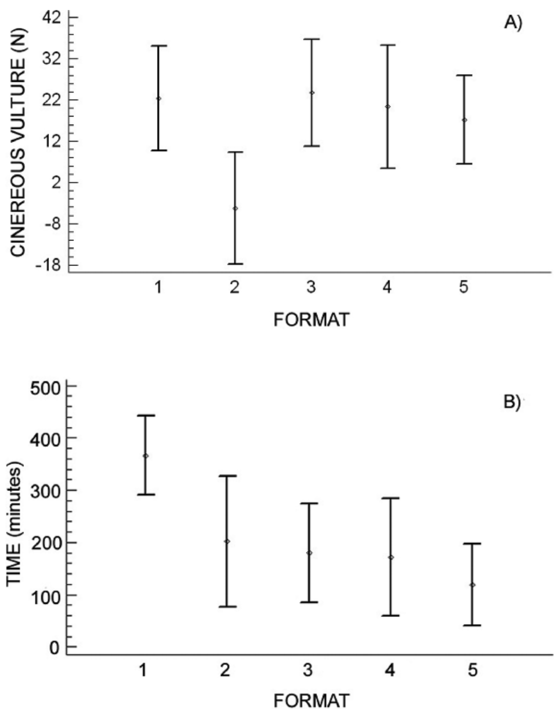

Figure 3. Number of cinereous vultures at carrion (A), and time difference between carrion delivery and vulture feeding (B) in relation to carrion format. 1) Whole carcass(es), 2) whole carcass(es) and piled up scraps, 3) whole carcass(es) and scraps scattered over a radius of up to 50 m, 4) piled up scraps, and 5) scraps scattered over a radius of up to 50 m.

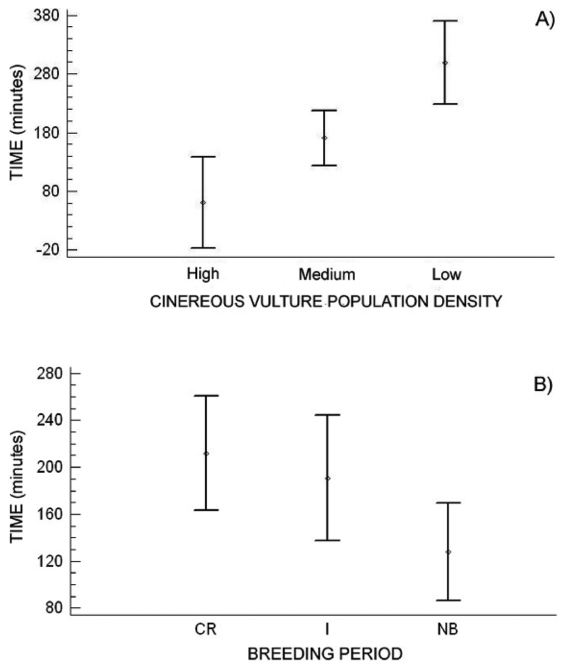

Figure 4. Time differences between carcass delivery and vulture feeding in relation to A) population density and B) breeding stage (CR: chick rearing, I: incubation, NB: non-breeding).

time that the cinereous vultures spent feeding depended significantly on the vulture density, and on the SPA*estate interaction, and marginally on the phenology (see Table 2 and Fig. 4).

With regard to age classes, the ratio of non-adult cinereous vultures to adult cinereous vultures was 1.77 ± 2.08 (range: 0-9, N=61). When we compared the proportion of different age classes in the carcasses through the breeding season, the adults visited the carcasses significantly more frequently during the chick-rearing period (F = 4.75, df = 2, 73, P = 0.001, Fig. 5), whereas the juveniles visited the carcasses more frequently during the non-breeding season (F = 4.81, df = 2, 73, P = 0.011). The subadults did not show significant differences among periods (F=0.30, df=2, 73, P=0.74; see Fig. 5).

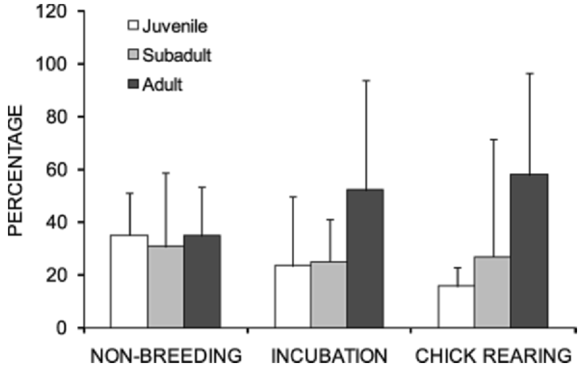

Figure 5. Variation in the ratio of age classes of the cinereous vultures present at the carcasses in accordance with breeding stage.

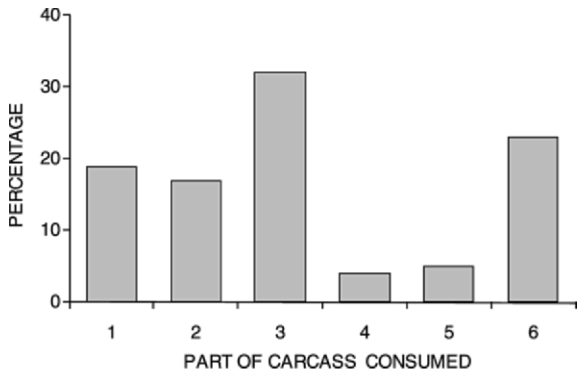

Figure 6. Categories of pieces ingested by cinereous vultures on different carcass parts: 1) all kinds of remains available at the carcass, 2) muscular pieces in a whole carcass, 3) scattered, medium-sized, muscular pieces, 4) entrails in a whole carcass, 5) entrails lying around a carcass, and 6) small peripheral scraps and tendons.

Carcass parts or remains most consumed among the six categories considered showed significant differences ($\chi^2=30.96$, df=5, P=0.004, N=99; Fig. 6). Cinereous vultures mostly fed on medium-sized muscular pieces and small remains and peripheral scraps, tendons etc. from the centre of the carcass, with observations of individuals eating entrails being rare, both at the carcasses and in the surrounding areas.

Discussion

The variables that explained the higher abundance and optimisation of the consumption of carrion by the cinereous vulture depended on several factors. Firstly, the quantity of biomass fed correlates directly with the total number of cinereous vultures that congregated as found for griffon vultures (Bosé & Sarrazin 2007), and also inversely with the time that elapsed before they began to eat. The average time the birds took to come to the carcass (214 minutes) was much lower than the time estimated for the cinereous vultures in the Caucasus (1,122 minutes, Gavashelishvili & McGrady 2006). This may be due to the proximity of the population nuclei and the high population density in our study area. In this respect, the high average number of individuals observed per carcass in our study (23.4 individuals), suggests that the use of conspecifics as cues to food location probably improves foraging success (Jackson et al. 2008). Secondly, the format of the food remains appeared to determine the access time and the total number of cinereous vultures attending the carcasses. Thus, when scattered, small or medium-sized remains were available, the cinereous vultures started eating more quickly than when large quantities of carrion were delivered to the same feeding station, which could favour other species such as the griffon vulture. This aspect could be of interest in relation to interspecific competition, dominance or ability to exploit food, which may explain differing selection of the type of remains (Hertel 1994, DeVault et al. 2003). Thus, the species would appear to be favoured by the consumption of medium-sized, relatively tough pieces, due to their external morphological adaptations to this type of food (König 1983).

The ratio of cinereous vultures compared with that of griffon vultures feeding at the carcass can constitute a measurement of the carrion consumption efficiency, since the griffon vulture will take more of the food at the carcass due to its numerical dominance and the species' adaptations to large numbers of individuals consuming big carcasses (König 1983). Our results suggested that the ratio of cinereous to griffon vultures is 1.29. These results suggest a ratio favourable for the cinereous vulture in comparison to all populations that live in the Iberian Peninsula, presumably because the study was carried out near large population nuclei of cinereous vultures, where their relative density is potentially greater (Carrete & Donázar 2005), and due to the abundance of wild ungulates and extensively reared livestock.

The species' breeding stage did not determine the number of individuals that came to a carcass, but was related to the duration of the consumption of the food. Moreover, when the age classes were separated in accordance with breeding stage, the results suggested that the juvenile age class exploited carcasses preferably during the winter, whereas adults increased their consumption during the chick-rearing period. This temporal segregation could be explained by the breeding population's greater energy requirements during the breeding season (Corbacho et al. 2007). In the case of the juvenile age class, the greater concentration during September-January could coincide with the post-fledging period before the start of juvenile dispersal. The fact that young birds have less foraging experience makes them more dependent on the movement of other scavengers such as the griffon vulture (see Jackson et al. 2008). Thus, the juveniles congregated in greater numbers during this period. During the spring and summer, the greater availability of trophic resources, milder weather and the start of the juvenile dispersal period could facilitate the birds leaving the breeding sites, and, therefore, could explain a lower number of birds at the feeding sites. In addition, these variables could have an impact on the time the cinereous vulture takes to search for food and its flight efficiency (Hiraldo & Donázar 1990).

Neither the number of nests in the radius around the feeding stations nor the plant cover appeared to determine the presence of cinereous vultures at carcasses. However, as central-place forager, the probabilities of obtaining food increase with the proximity to breeding nuclei (Carrete & Donázar 2005). In addition, the cinereous vulture is better adapted than other vulture species to detect prey in more overgrown areas (e.g. scrub, wasteland, 'dehesas' (pastureland with holm and cork oaks, grazed by livestock) and grazing land), although at all times this is related to prey availability (Carrete & Donázar 2005, Costillo 2005). The type of food consumed was mainly small or medium-sized items, tough or relatively tough pieces (e.g. muscles, tendons and skins), preferably scattered and not concentrated in one place (König 1983).

In summary, the cinereous vulture benefits from carcasses being delivered with significant biomass, broken up into small and medium-sized pieces that are scattered and not concentrated in one place. In order to increase the cinereous vulture's use of this resource, the delivery of a higher number of separate pieces of carrion would favour the birds' presence. In this respect, the relatively tough, medium-sized remains and pieces of muscle and tissue, and the small, scattered pieces are those that the cinereous vulture eats most efficiently.

Implications for conservation

The management of the feeding of threatened avian scavengers is a major challenge currently facing conservationists (Deygout et al. 2009, Donázar et al. 2009a). In endangered species, supplementary food has been proved useful for increasing pre-adult survival (Oro et al. 2008) and breeding parameters (González et al. 2006, López-Bao et al. 2008). Vulture restaurants also may help to reduce vulture mortality by providing safe food (not contaminated by pesticides and confirmed not to contain transmissible agents of zoonosis or livestock diseases because of the human control of the deliveries, see Gilbert et al. 2007, Margalida et al. 2008b, Hernández & Margalida 2009). However, supplementary feeding can also have negative effects (see Robb et al. 2008). The creation of supplementary feeding stations in which large quantities of food are delivered and pile up, does not favour the most threatened avian scavengers. In this respect, it can influence the spatial distribution of the breeding population (Margalida et al. 2008a). It can also attract facultative scavengers, which could predate on species living in the surrounding area (Cortés-Avizanda et al. 2009), and thereby have a detrimental effect on fecundity (Carrete et al. 2006). Moreover, recent studies show a potential increase in levels of antibiotics in the birds' blood as a consequence of feeding on carcasses from intensive farming feeding stations (Lemus et al. 2008). Thus, this procedure of feeding should be based on rigorous information that allows the pros and cons of this type of management to be evaluated. In this regard, managers must assess what type of remains the cinereous vulture prefers, how the remains should be laid out in order to optimise their consumption, and how to discourage less endangered species such as the griffon vulture using this resource. However, there is still a need for more studies about these subjects in the scientific literature, which can provide guidelines regarding the use of feeding stations for the whole scavenger raptor community.

When managing supplementary feeding stations for cinereous vultures, managers should focus on quantity, format and the number and dispersal of the pieces, in order to optimise the birds' consumption of the carrion. Moreover, carrion should be mainly supplied during the chick-rearing period within the breeding areas, in order to increase the species' productivity and avoid the above-mentioned negative side effects (Deygout et al. 2009). These results suggest that the feeding of the species should be based on the appropriate sustainable measures, such as carcasses from extensively reared livestock, which occur more heterogeneously on spatial and temporal levels (Bosé & Sarrazin 2007, Margalida et al. 2007).

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References

BirdLife International 2004: Birds in Europe. Population estimates, trends and conservation status. - BirdLife Conservation Series 12, BirdLife International, 374 pp.

BirdLife International 2009: *Aegypius monachus*. - IUCN Red List of Threatened Species, Version 2009.2, IUCN 2009. Available at: http://www.iucnredlist.org (Last accessed on 20 December 2009).

Bosé, M. & Sarrazin, F. 2007: Competitive behaviour and feeding rate in a reintroduced population of griffon vultures *Gyps fulvus*. - Ibis 149: 490-501.

Camiña, A. & Montelío, E. 2006: Griffon vulture *Gyps fulvus* food shortages in the Ebro valley (NE Spain) caused by regulations against Bovine Spongiform Encephalopathy (BSE). - Acta Ornithologica 41: 7-13.

Carrete, M. & Donázar, J.A. 2005: Application of central-place foraging theory shows the importance of Mediterranean dehesas for the conservation of cinereous vulture *Aegypius monachus*. - Biological Conservation 126: 582-590.

Carrete, M., Donázar, J.A. & Margalida, A. 2006: Density-dependent productivity depression in Pyrenean bearded vultures: implications for conservation. - Ecological Applications 16: 1674-1682.

Corbacho, C., Costillo, E. & Perales, A.B. 2007: La alimentación del buitre negro. - In: Moreno-Opo, R. & Guil, F. (Eds.); Manual de gestión del hábitat y de las poblaciones de buitre negro en España. Dirección General para la Biodiversidad, Ministerio de Medio Ambiente, Madrid, Spain, pp. 179-196. (In Spanish).

Cortés-Avizanda, A., Selva, N., Carrete, M., Serrano, D. & Donázar, J.A. 2009: Carcasses increase the probability of predation of groundnesting birds: a caveat regarding the conservation value of vulture restaurants. - Animal Conservation 12: 85-88.

Costillo, E. 2005: Biología y Conservación de las poblaciones de buitre negro *Aegypius monachus* en Extremadura. - PhD thesis, Universidad de Extremadura, 241 pp. (In Spanish).

Costillo, E., Corbacho, C., Morán, R. & Villegas, A. 2007: The diet of the black vulture *Aegypius monachus* in response to environmental changes in Extremadura (1970-2000). - Ardeola 54: 197-204.

De la Puente, J., Moreno-Opo, R. & Del Moral, J.C. 2007: El buitre negro en España. Censo nacional (2006). (In Spanish with an English summary: The black vulture in Spain. National Census 2006). - SEO/BirdLife, Madrid, Spain, 113 pp.

DeVault, T.L., Rhodes, O.E. & Shivik, J.A. 2003: Scavenging by vertebrates: behavioural, ecological, and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. - Oikos 102: 225-234.

Deygout, C., Gault, A., Sarrazin, F. & Bessa-Gomes, C. 2009: Modelling the impact of feeding stations on vulture scavenging efficiency. - Ecological Modelling 220: 1826-1835.

Donázar, J.A. 1993: Los buitres ibéricos. - In: Reyero, J.M. (Ed.), Biología y conservación. Madrid, Spain, 253 pp. (In Spanish).

Donázar, J.A., Margalida, A. & Campión, D. (Eds.) 2009a: Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology. - Munibe 29 (Suppl.), Sociedad de Ciencias Aranzadi, Donostia, Spain, 551 pp.

Donázar, J.A., Margalida, A., Carrete, M. & Sánchez-Zapata, J.A. 2009b: Too sanitary for vultures. - Science 326: 664.

García de Francisco, J.M. & Moreno-Opo, R. 2009. The cattle carcasses management today: is there enough flexibility to deal new conservation strategies?. - In: Donázar, J.A., Margalida, A. & Campión, D. (Eds.); Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology. Munibe 29 (Suppl.), Sociedad de Ciencias Aranzadi, Donostia, pp. 452-509.

Gavashelishvili, A. & McGrady, M.J. 2006: Geographic information system-based modelling of vulture response to carcass appearance in the Caucasus. - Journal of Zoology 269: 365-372.

Gilbert, M., Watson, R.T., Ahmed, S., Asim, M. & Johnson, J.A. 2007: Vulture restaurants and their role in reducing diclofenac exposure in Asian vultures. - Bird Conservation International 17: 63-77.

González, L.M., Margalida, A., Sánchez, R. & Oria, J. 2006: Supplementary feeding as an effective tool for improving breeding success on Spanish imperial eagle *Aquila adalberti*. - Biological Conservation 129: 477-486.

Hernández, M. & Margalida, A. 2008: Pesticide abuse in Europe: effects on the cinereous vulture *Aegypius monachus* in Spain. - Ecotoxicology 17: 264-272.

Hernández, M. & Margalida, A. 2009: Poison-related mortality effects in the endangered Egyptian vulture *Neophron percnopterus* population in Spain. - European Journal of Wildlife Research 55: 415-423.

Hertel, F. 1994: Diversity in body size and feeding morphology within past and present vulture assemblages. - Ecology 75: 1074-1084.

Hiraldo, F. 1976: Diet of the black vulture *Aegypius monachus* in the Iberian Peninsula. - Doñana, Acta Vertebrata 3: 19-31.

Hiraldo, F. & Donázar, J.A. 1990: Foraging time in the Cinereous Vulture *Aegypius monachus*: seasonal and local variations and influence of weather. - Bird Study 37: 128-132.

Jackson, A.L., Ruxton, G.D. & Houston, D.C. 2008: The effect of social facilitation on foraging success in vultures: a modelling study. - Biology Letters 4: 311-313.

Jones, G.G. 2004: Conservation management of endangered birds. - In: Sutherland, W.J., Newton, I. & Green, R.E. (Eds.); Bird Ecology and Conservation: a Handbook of Techniques, Oxford University Press, Oxford, UK, pp. 269-301.

König, C. 1983: Interspecific and intraspecific competition for food among Old World vultures. - In: Wilbur, S.R. & Jackson, J.A. (Eds.); Vulture biology and management, University of California Press, Berkeley, USA, pp. 153-171.

Lemus, J.A., Blanco, G., Grande, J., Arroyo, B., García-Montijano, M. & Martínez, F. 2008: Antibiotics threaten wildlife: circulating quilonone residues and disease in avian scavengers. - Plos One 3: e1444.

López-Bao, J.V., Rodríguez, A. & Palomares, F. 2008: Behavioural response of a trophic specialist, the Iberian lynx, to supplementary food: patterns of food use and implications for conservation. - Biological Conservation 141: 1857-1867.

Margalida, A., Bertran, J. & Heredia, R. 2009: Diet and food preferences of the endangered Bearded vulture *Gypaetus barbatus*: a basis for their conservation. - Ibis 151: 235-243.

Margalida, A., Donázar, J.A., Bustamante, J., Hernández, F. & Romero-Pujante, M. 2008a: Application of a predictive model to detect long-term changes in nest-site selection in the Bearded Vultures: conservation in relation to territory shrinkage. - Ibis 150: 242-249.

Margalida, A., García, D. & Cortés-Avizanda, A. 2007: Factors influencing breeding density of Bearded Vultures, Egyptian Vultures and Eurasian Griffon Vultures in Catalonia (NE Spain): management implications. - Animal Biodiversity & Conservation 42: 189-200.

Margalida, A., Heredia, R., Razin, M. & Hernández, M. 2008b: Sources of variation in mortality of the Bearded vulture *Gypaetus barbatus* in Europe. - Bird Conservation International 18: 1-10.

Moreno, J. 2002: La evolución de las estrategias vitales. - In: Soler, M. (Ed.); Evolución. La base de la Biología, Manuel Soler Editor, pp. 159-176. (In Spanish).

Moreno-Opo, R. 2007: El buitre negro. - In: Moreno-Opo, R. & Guil, F. (Eds.); Manual de gestión del hábitat y de las poblaciones de buitre negro en España. Dirección General para la Biodiversidad, Ministerio de Medio Ambiente, Madrid, Spain, pp. 25-45. (In Spanish).

Moreno-Opo, R., San Miguel, A. & Camiña, A. 2007: Ganadería y buitre negro. - In: Moreno-Opo, R. & Guil, F. (Eds.); Manual de gestión del hábitat y de las poblaciones de buitre negro en España. Dirección General para la Biodiversidad, Ministerio de Medio Ambiente, Madrid, Spain, pp. 200-223. (In Spanish).

Oro, D., Margalida, A., Carrete, M., Heredia, R. & Donázar, J.A. 2008: Testing the goodness of supplementary feeding to enhance population viability in and endangered vulture. - Plos One 3: e4084.

Ostfeld, R.S. & Keesing, F. 2000: Pulsed resources and community dynamics of consumers in terrestrial ecosystems. - Trends in Ecology & Evolution 15: 232-237.

Robb, G.N., McDonald, R.A., Chamberlain, D.E. & Bearhop, S. 2008: Food for thought: supplementary feeding as a driver of ecological change in avian populations. - Frontiers in Ecology Environment 6: 476-484.

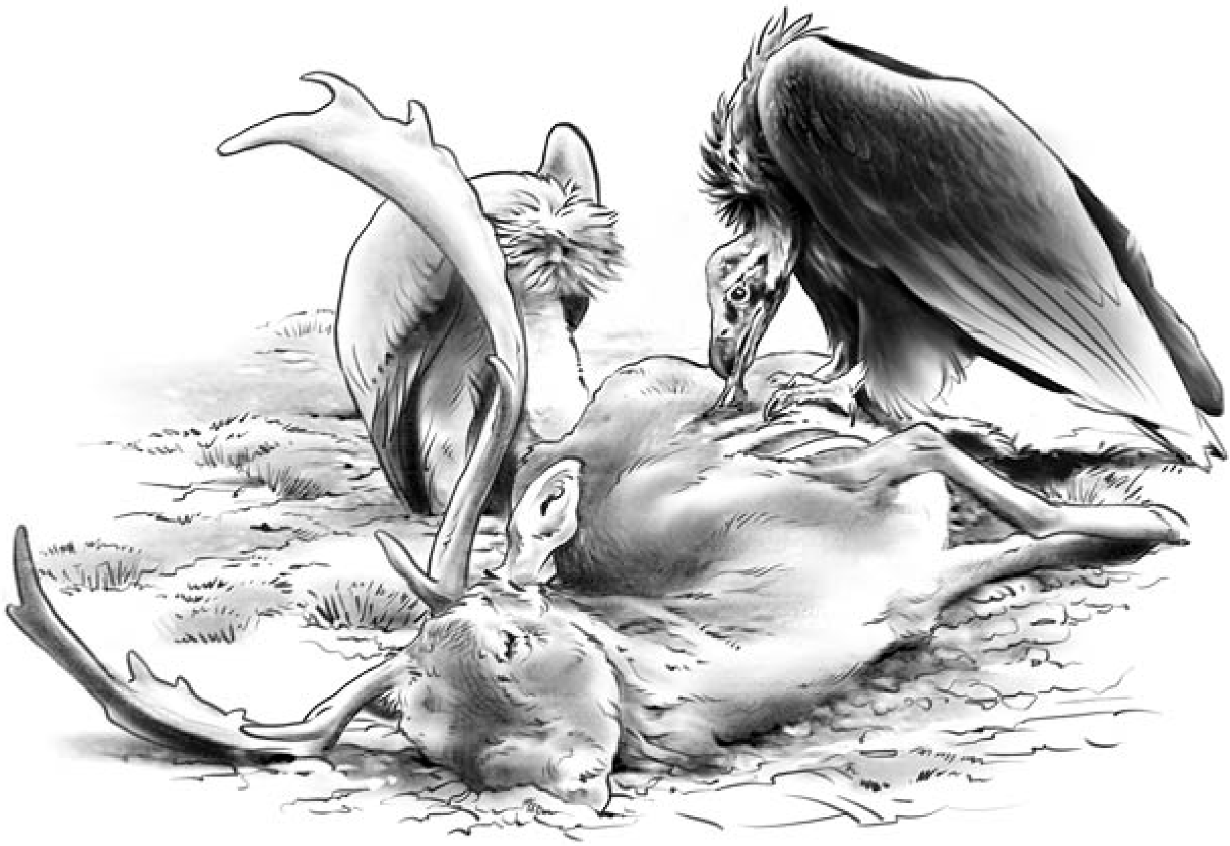
Sánchez, J.J. 2004: Buitre negro *Aegypius monachus*. - In: Madroño, A., González, C. & Atienza, J.C. (Eds.); Libro Rojo de las Aves de España, SEO/BirdLife-Dirección General para la Biodiversidad, Madrid, Spain, pp. 170-171. (In Spanish).

Tella, J.L. 2001: Action is needed now or BSE crisis could wipe out endangered birds of prey. - Nature 410: 408.

Vasilakis, D., Poizaridis, K. & Elorriaga, J. 2006: Breeding season range use of a Eurasian Black Vulture *Aegypius monachus* population in Dadia National Park and the adjacent areas, Thrace, NE Greece. - In: Houston, D.C. & Piper, S.C. (Eds.); Proceedings of the International Conference on Conservation and Management of Vulture Populations, Natural History Museum of Crete/WWF Greece, Thessaloniki, Greece, pp. 127-137.

GESTIÓN DEL HÁBITAT DE REPRODUCCIÓN Y ALIMENTACIÓN

MANAGEMENT OF BREEDING AND FEEDING HABITAT



CAPÍTULO-CHAPTER
5

Reconciling the conservation of endangered species with economically important anthropogenic activities: interactions between cork exploitation and the cinereous vulture in Spain

by

Antoni Margalida, Rubén Moreno-Opo, Beatriz Arroyo & Ángel Arredondo
(2011)

Animal Conservation **14: 167-174**

Compatibilizando la conservación de especies amenazadas con actividades humanas económicamente importantes: interacciones entre descorche y buitre negro en España

La limitación de actividades que generan molestias alrededor de lugares de nidificación de especies protegidas no siempre es posible, en tanto estas actividades tienen una elevada importancia desde el punto de vista económico y, además, generan efectos positivos sobre la protección del hábitat de dichas especies protegidas. La búsqueda de soluciones que permitan compatibilizar la explotación comercial de recursos naturales con la conservación de la biodiversidad es un asunto de gran importancia para gestores y responsables políticos. Este es el caso del buitre negro, que cría en numerosos bosques de alcornoques, y el descorche, una actividad socioeconómica tradicional desarrollada en distintos países mediterráneos que resulta clave para el mantenimiento de este importante hábitat natural. Se estudiaron los efectos de este aprovechamiento humano sobre el comportamiento y el éxito reproductivo de buitres negros reproductores en España. Para los adultos, la probabilidad de abandono del nido dependió de la distancia de los trabajadores al nido y el nivel de ruido que éstos generaron; las actuaciones a menos de 500m del nido provocaron una mayor probabilidad de abandono del nido por los adultos, en el caso de que el nivel de ruido fuera medio o alto. Ni el número de personas trabajando ni el uso de maquinaria para la explotación resultaron tener influencia sobre el abandono del nido. Las parejas reproductoras situadas en las zonas donde se desarrolló el descorche y, por tanto, expuestas a molestias humanas, tuvieron un 20% menos de éxito reproductivo que aquellas de la misma colonia no expuestas a dichas molestias. Si la aplicación de zonas protegidas-tampón (buffer zones) no es posible, como es el caso por los grandes pérdidas económicas que supondrían, se recomiendan distintas alternativas provenientes de nuestros resultados que minimizan el impacto de estas actividades, sobre todo la reducción del nivel de ruido durante la extracción del corcho. Estudios observacionales como éste contribuyen a la comprensión de la magnitud de distintas amenazas para especies amenazadas y al hallazgo de soluciones alternativas que armonicen conservación y desarrollo económico.

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Reconciling the conservation of endangered species with economically important anthropogenic activities: interactions between cork exploitation and the cinereous vulture in Spain

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Keywords
buffer zone; cork harvesting; economic anthropogenic activity; human disturbance; vulture conservation.

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Abstract

Limitation of disturbing activities around the breeding areas of protected species is not always possible, if these activities are economically important and have, in addition, positive effects on protecting the habitats of those protected species. Searching for optimal solutions making commercial exploitation of natural resources compatible with biodiversity conservation is thus of concern to managers and policy makers. This is the case of the cinereous vulture *Aegypius monachus*, breeding primarily in cork-oak woodland, and cork exploitation, a traditional socio-economic activity carried out in several Mediterranean countries, and critical for the maintenance of this important habitat. We studied the effects of this anthropogenic activity on the behaviour and breeding success of breeding cinereous vultures in Spain. For the adults, the probability of nest abandonment was dependent on the distance of workers from the nest and the level of noise; activities within 500 m from the nest were likely to cause abandonment of the nest by adults, if the level of noise was intermediate or loud. Neither the size of the working group nor the use of machines *per se*, had any effect on the probability of nest abandonment. Pairs in an area of the colony exposed to intrusive anthropogenic activity had 20% lower breeding success than those in the same colony that were not exposed to these disturbances. If the application of buffer zones of 500 m is not possible (as is likely given the economic losses involved), several alternatives are recommended based on our results to minimize the impact of these activities, in particular to diminish the noise level of cork extraction activities. Observational studies like this help understanding the magnitude of the problem and finding alternative solutions for harmonizing conservation and economic development.

Introduction

The relationships between the development of human activities and wild animal species are of concern to managers and policy makers searching for alternative solutions making conservation compatible with anthropogenic activities (Young *et al*., 2005; Preisler, Ager & Wisdom, 2006). Human disturbance and its effects on the behaviour or breeding success of birds are a widely studied subject in several avian species (Blumstein *et al*., 2005; Gill, 2005; Langston *et al*., 2007), including threatened raptors (Steidl & Anthony, 2000; Arroyo & Razin, 2006; González *et al*., 2006; Zuberogoitia *et al*., 2008). The conservation measure most often used by managers and conservationists to avoid disturbance is the establishment of spatial and temporal buffer zones around potentially sensitive areas (e.g. breeding sites), where the disturbing activities are limited or prohibited. The radius for these buffer areas usually depends on the sensitivity of the species, and may be calculated through the observations of distances at which an activity produces alert behaviour or nest abandonment (Richardson & Miller, 1997; Fernández-Juricic *et al*., 2005; Whitfield, Ruddock & Bullman, 2008). The establishment of buffer zones is generally easily regulated in those cases when the disturbing activities are leisure activities, such as hunting or ecotourism (see Richardson & Miller, 1997; Guil & Moreno-Opo, 2008). However, in the case of activities with strong economic interests (industrial activities such as the construction of highways and roads or forest-related activities such as logging), the implementation of conservation measures may be more difficult (Donázar *et al*., 2002; Bautista *et al*., 2004; Speziale, Lambertucci & Olsson, 2008). Additionally, the economic interests associated with the same activities that may be disturbing to the birds may help to maintain the habitat on which the birds themselves depend. This may be the case with the exploitation of forest products causing disturbance to the fauna in the area in which the activity is carried out. Harmonizing conservation and economic development can pose a challenge for managers and necessitates appropriate and objective research studies leading to solutions making anthropogenic activities compatible with the conservation of a threatened species.

An example of this dilemma is the case of the cinereous vulture *Aegypius monachus* and cork harvesting, a traditional socio-economic activity carried out in several Mediterranean countries including Spain and Portugal, as well as Morocco, France, Italy and Algeria. The cinereous vulture is a species considered Near Threatened by IUCN (BirdLife International, 2008) and the Spanish population (*c*. 1845 pairs) represents 98% of the European population and between 18 and 25% of the world population (De la Puente, Moreno-Opo & Del Moral, 2007). This species breeds frequently in cork oak *Quercus suber* trees in Spain. Cork harvesting generates an annual turn over of €1.5 billion and results in the direct and indirect creation of around 100 000 jobs (WWF, 2006), but it is considered to be one of the main causes of disturbance to the cinereous vulture during its breeding period, because this activity is carried out in June–July, while chicks are being reared (Moreno-Opo & Arredondo, 2007). During this period, it is essential that the adults provide their chicks with shade in order to protect them from direct sunlight, which could lead to dehydration of the chicks (Donázar, 1993; Moreno-Opo & Arredondo, 2007). The study of the interaction of this activity and any possible disturbance that it may create and its impact on breeding cinereous vultures might offer information that could enable possible solutions to be found.

In this paper, we examine the effects of cork harvesting on the cinereous vulture's behaviour during the breeding season and on their breeding success. We discuss the results to assess whether and in what circumstances this anthropogenic activity may be compatible with the conservation of this endangered vulture and, more generally, discuss the use of observational studies as a tool in finding optimal solutions for harmonizing conservation and economic development.

Methods

Study area

Cork harvesting and its impact on the cinereous vulture's breeding was monitored in June 2005 in the Umbría de Alcudia colony (Ciudad Real, Spain, Fig. 1), an 11 115 ha sector containing 99 breeding pairs. Within this area, cork extraction in 2005 occurred in a 3200 ha area, containing 51 pairs (Fig. 2).

The habitat in the study area consists of mature Mediterranean landscapes, made up of arboreal species such as the cork oak *Q. suber*, holm oak *Quercus ilex*, prickly juniper *Juniperus oxycedrus* and the strawberry tree *Arbutus unedo*, with a well-developed shrub cover, on slopes with an incline of 25–45% and at altitudes ranging between 736 and 1115 m.

The cork harvest

Cork harvesting is a forestry activity that consists of the removal of the bark from cork oaks. Each tree is harvested every 9–10 years. The initial prospecting activity and the subsequent harvest are carried out between May and mid August with the optimum time for cork extraction (according to the physiological state of the tree) being between June and mid July (Borges, Oliveira & Costa, 1997; Pereira,

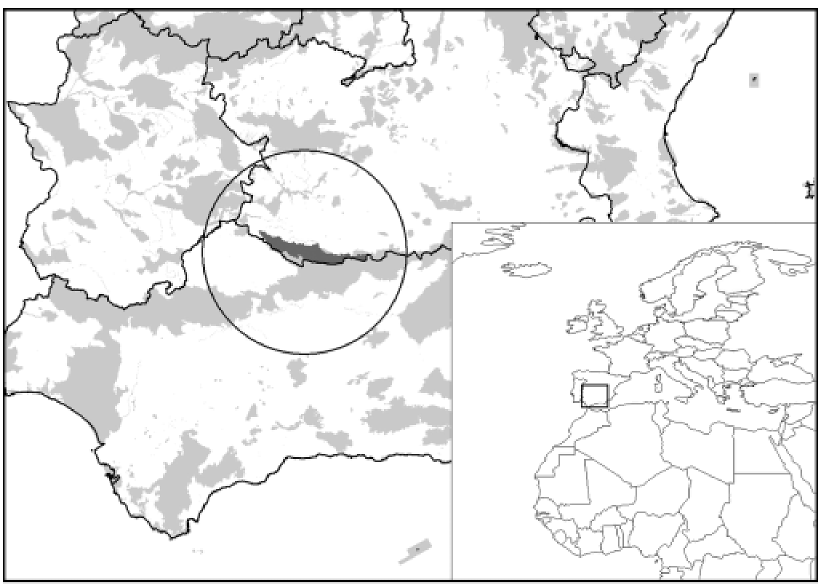

Figure 1 General location of the study area in Spain. Special protected area (SPA) in which the breeding colony of cinereous vultures *Aegypius monachus* studied is found, highlighted in dark grey. In light grey appears the whole Natura 2000 network in the Southern mid Spain.

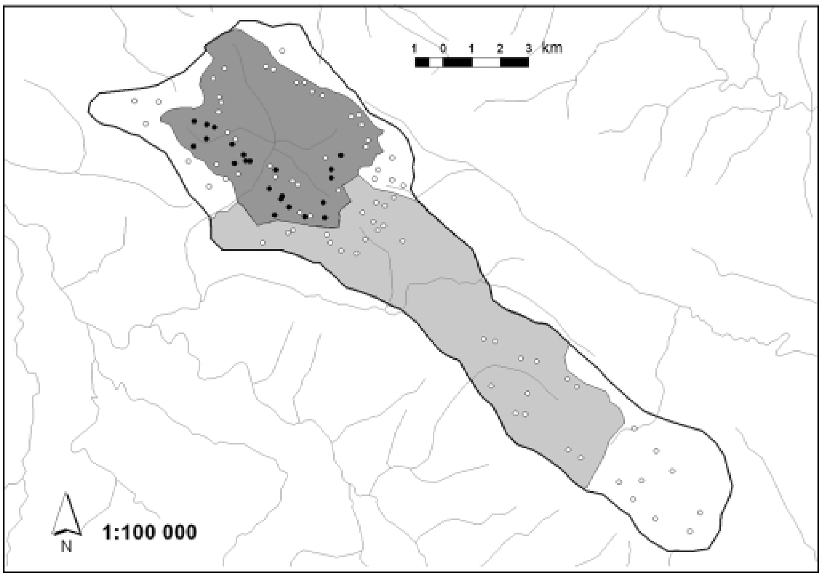

Figure 2 Location of the cinereous vulture *Aegypius monachus* breeding pairs in the studied colony in 2005. Dark grey corresponds to the cork harvesting area ($n=51$ nests), while light grey is the control area for breeding success analysis ($n=28$ nests). Black dots: nests in which vulture behaviour was studied in relation to cork-harvesting activity ($n=22$); white dots: rest of nests of breeding cinereous vultures ($n=77$). Black line: outline of the breeding colony; grey lines: river network.

2007). The harvest is carried out between 07:00 AM and 02:30 PM by teams of 15–30 workers, who remove the cork with manual tools and then transport it using animal transport and/or vehicles.

Field procedures

First, we conducted meetings with the managers of the cork harvesting to know in advance the areas and timing where harvesting works would occur. A total of 12 days in June 2005 were spent monitoring the impact of cork harvesting on 22 cinereous vulture nests with chicks, with a total of 122 observations. The minimum distance between neighbouring pairs was 150 m. Observers were placed at >800 m from the areas with cork extraction activity, usually in the opposite slopes to where work was carried out. Each observation day corresponded to a different harvesting event. On average, we monitored simultaneously two nests daily (range 1–5, $n = 14$). Observations were carried out with $\times$20–60 spotting scopes. The visible reactions of breeding individuals and chicks were noted. These were divided into three categories for the adults: *No reaction*, when the bird displayed no apparent change in behaviour; *alert reaction*, when the bird stood up in the nest with its head outstretched and looked in the direction of the human activity, but did not leave the nest; and *flight reaction*, when the adult bird left the nest. In the third case, the minimum time the adults were absent from the nest was noted (in some cases the observation ended before the adult returned to the nest). In the case of the chicks, we described chick behaviour as *no reaction* or *alert reaction* as above. In each case, we also noted the following variables: (1) the distance between people and the nest, noted by the positions in aerial photographs and the subsequent measurement of distances using GIS computer programs; (2) the noise level as perceived by the observers (and thus, theoretically, as perceived by the vultures), considering this to be *zero* (no voices), *slight* (when only a few voices were heard sporadically), *moderate* (when voices were heard often, but not constantly or loudly) and *loud* (when generalized voices were heard constantly and at a loud volume), independently; (3) the number of people present; (4) the presence or absence of machinery.

To evaluate whether cork extraction activities influenced breeding success, we compared two areas of the colony in 2005 (Fig. 2). The two areas correspond to two different private estates, with different management regimes, but have similar habitat characteristics and similar vulture breeding densities. In one area, containing 51 breeding pairs, cork harvesting occurred in 2005. The other one (hereafter called 'control area'), containing 28 pairs, had no cork harvesting activities in 2005. Breeding success was monitored in both areas with the same methodology and survey effort. Observations of all nests in the colony were periodically conducted each fortnight; we noted the breeding status in each visit (nest occupied or unoccupied and, in the former case, incubation, chick present, adult/s presence/absence, abandonment or failure, chick fledged). For each area, we calculated the breeding success (number of pairs in which a chick fledged divided by total number of pairs with clutches) and the chick mortality rate (number of nests in which the chick died divided by the total number of nests with hatched eggs). Cinereous vultures rear a maximum of one chick per breeding event (Hiraldo, 1983).

Statistical analyses

We carried out generalised linear mixed models to analyse vulture behaviour. We included nest, date (i.e. harvesting event) and their interaction as random variables to account for the lack of independence of observations carried out the same day for different nests and different observations (in the same or different dates) for the same nest.

We first analysed the probability that the nestling would be alarmed in relation to various attributes of the activity. As explanatory variables we included distance to the nest, number of people involved, level of noise (because of the sample size for this analysis, we combined the zero and slight noise categories), whether vehicles and machines were involved or not, the interaction between distance and noise, the interaction between distance and presence of machines and the interaction between distance and number of people.

In analysing the effect of cork harvesting work on the behaviour of the adults, we were particularly interested in evaluating the probability of reactions that may have a potential effect on reproductive success. We thus analysed the probability of nest abandonment (and lumped observations when adults showed ‘no reaction’ or ‘alert reaction’ for analyses). The initial model was the same as that described above for nestlings.

The dependent variables were fitted with a binomial error, and a log link function. Backward selection was used to identify the most parsimonious model. Type III results are presented.

Finally, in order to observe the impact of cork harvesting on the breeding parameters of cinereous vultures, we tested whether breeding success and chick mortality rate varied between the subsections of the colony that were subjected to human disturbance ($n = 51$ nests) and the control area ($n = 28$ nests), with Fisher’s exact tests.

Results

Effect of cork harvesting on vulture behaviour

The probability of nestlings being alarmed depended only on distance between the activity and the nest, and on the level of noise ($F_{1,102} = 15.44$, $P = 0.0001$ for distance, $F_{2,102} = 4.99$, $P = 0.009$ for level of noise). Activities with a high level of noise occurring within 500 m from the nest had a high probability of alarming the nestlings (Fig. 3a). If the level of noise was low, nestlings were alarmed only when the activities were closer to the nest (Fig. 3a).

For the adults, the probability of nest abandonment was also only dependent on distance from the nest, and there was a near significant effect on the level of noise ($F_{1,78} = 10.66$, $P = 0.002$ for distance; $F_{1,78} = 2.55$, $P = 0.08$ for level of noise). Any activity within 500 m from the nest, if the level of noise was intermediate or loud, was highly likely to cause abandonment of the nest by adults (Fig. 3b). The average observed flight distance was 220.21 ± 153.8 m (range 10–600, $n = 23$). Average observed alert distance was 332.2 ± 174.2 m (range 50–700, $n = 39$). When adults flew from the nest, the average observed time of nest abandonment was 132 ± 85 min (range 12–330 min, $n = 22$).

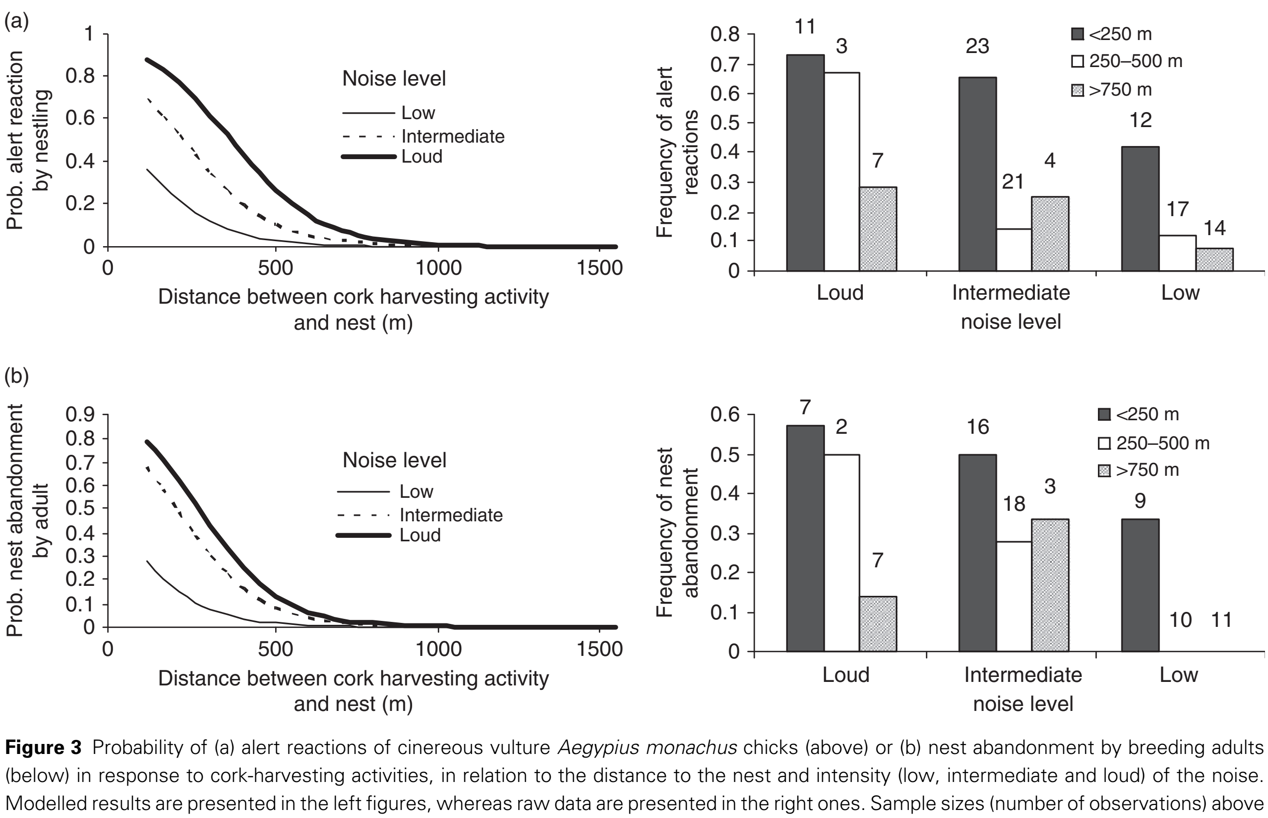

Figure 3 Probability of (a) alert reactions of cinereous vulture *Aegypius monachus* chicks (above) or (b) nest abandonment by breeding adults (below) in response to cork-harvesting activities, in relation to the distance to the nest and intensity (low, intermediate and loud) of the noise. Modelled results are presented in the left figures, whereas raw data are presented in the right ones. Sample sizes (number of observations) above bars.

Influence on breeding success

The breeding success of pairs in the area with cork harvesting activities was 0.55 ± 0.50 chicks per pair with clutches ($n = 51$), whereas in the control area it was 0.75 ± 0.44 ($n = 28$). Differences approached statistical significance (Fisher's exact test $P = 0.06$).

Taking into account only the pairs where hatching was observed, in the cork harvesting area chick mortality was 24.6% ($n = 37$ nests with hatching), whereas in the control area this percentage was three times lower, 8.7% ($n = 23$), although the differences were not statistically significant (Fisher's exact test $P = 0.12$). In the cork harvesting area, two chicks died dehydrated at age 29 and 32 days old as a consequence of nest abandonment by the adults directly related to the cork harvesting activity, as showed by necropsies conducted in official veterinarian centres. In a third nest, cork harvesting activity led to nest abandonment for 2 consecutive days and the chick was rescued the second day at the age of 26 days with symptoms of dehydration.

Discussion

Our observations indicate that cork extraction activities resulted in a high likelihood that breeding cinereous vultures would leave the nest for long periods and in lower breeding success. However, they also suggest that the overall impact of this loss at the population level may be lower than that arising from stopping cork harvesting, indicating how to best minimize this impact. We discuss these results below, as well as the conflict between maintenance of economic activities, habitat and wildlife conservation.

Cork harvesting and cinereous vulture conservation

Nest abandonment by breeding cinereous vultures during the incubation or early chick-rearing periods is rare because of the high temperatures and the consequent risk of chick dehydration (Hiraldo, 1983). The higher probability of nest abandonment during cork extraction observed here is thus likely to have strong impacts on the nestlings and, indeed, our data indicate that nests exposed to this activity had a chick mortality three times higher than those in a control area (although this difference was not statistically significant). Our results also suggest that the elimination of the disturbance could lead to a 20% increase in successful nests. The life history of a long-lived species such as this is characterized by low fecundity and high survival rate (Hiraldo, 1983), and adult survival is the most sensitive parameter affecting variations in population dynamics (e.g. Meretsky *et al.*, 2000; Oro *et al.*, 2008). Thus, it could be argued that the global effects of disturbance in the vulture populations might be negligible (particularly bearing in mind that this activity is carried out on each tree every 9–10 years, and hence, the activities affect only a fraction of the breeding population each year). Nevertheless, as yet, no demographic models have been constructed to measure the effect of this loss and, in any case, this decrease in productivity could be an added detrimental factor to already threatened populations. In addition, it could be useful to measure the non-lethal physiological stress effects of disturbance (Gill, Norris & Sutherland, 2001; Holmes, Giese & Kriwoken, 2005), which may have consequences beyond the nest stage. A simple method to measure the stress level would be to analyse corticosterone levels in chick feathers collected in the nests (Bortolotti *et al.*, 2008).

Our results showed that activities occurring <500 m from the nest had a $>25\%$ probability of adults abandoning the nest. This suggests that cork harvesting activities should be minimized within 500 m of active nests in order to avoid disturbance, and the subsequent decrease in breeding success. However, each kilo of cork generates a turnover of €1.43 (MMAMRM, 2009), and in the area of 500 m around a nest, the amount of money lost could be as great as €67 650. Moreover, cork that is not removed at the optimal time decreases in unit value and thus could be lost for future generations. A delay in cork extraction until August, when nestlings have fledged, is not a valid option because extraction outside the optimal physiological time for the tree also has added costs (Borges *et al.*, 1997; Pereira, 2007). Ultimately, the exploitation of the cork oaks allows these forests to be conserved, due to their economic yield leading to their protection (WWF, 2006), and biodiversity conservation necessitates not only the preservation of species but also their habitats (Behera, Kushwaha & Roy, 2005). This creates a dichotomy from a conservation standpoint, because cork-harvesting activity is interlinked to the conservation of the cinereous vulture.

A cost–benefit analysis suggests, as described above, that the costs of reducing cork harvesting activities are much greater than the benefits (in terms of increased productivity) for the cinereous vultures. Finding a compromise that maintains this economic activity while minimizing the detrimental effects on this endangered species should be considered. Our data show that, in addition to distance, the noise level also determined the reaction of species toward disturbance, a factor that has been shown to modify the behaviour of other bird species that come in contact with humans (Bowles, 1995; Bautista *et al.*, 2004; Arroyo & Razin, 2006). The distance at which there was a high probability of nest abandonment during cork extraction was greatly reduced if the activity was silent, or almost silent. Hence, simply diminishing the noise level would promote the conservation of such umbrella Mediterranean raptor species. In addition, aspects such as carrying out the activities during hours of cooler temperatures, not prolonging activities affecting the same nests by more than one consecutive day or carrying out work on a slope below the nest such that activity can be observed by adults from above, would strongly minimize the impact of these activities. Thus, supervision of cork-extraction activities by teams of technicians trained in advising workers and in taking action towards rescuing birds subjected to a prolonged absence of the parents is advisable.

Reconciling conservation of endangered species with economically important anthropogenic activities

The conservation of threatened species is a crucial challenge in the current context of biodiversity loss, and there are social and international objectives for halting this loss (Butchart *et al.*, 2010; Marton-Lefèvre, 2010). However, it is increasingly recognized that the creation of protected areas alone is not always sufficient for the preservation of biodiversity. Finding adapted, alternative solutions as in our case may result in more effective conservation. Thus, it is extremely important to integrate the management of protected areas with the human activities and land use occurring in their surroundings (Sergio *et al.*, 2005), particularly for species living in habitats dominated and conditioned by human activities. In most cases (at least within populated Europe), conservation of threatened species needs to be compatible with human activities if conservation programmes aim to be sustainable (Margalida *et al.*, 2010). Additionally, the economic benefits arising from human activities may help to preserve the ecosystems in which they occur. For example, the economic benefit from red grouse *Lagopus lagopus scoticus* hunting helps to maintain the ecologically important heather moorland habitat (Thompson *et al.*, 1995). Otherwise, human activities may cause negative effects on wildlife, such as disturbance, causing disruption of normal breeding behaviour or even breeding failure in wildlife (Blumstein *et al.*, 2005; Arroyo & Razin, 2006; González *et al.*, 2006; Langston *et al.*, 2007; Zuberogoitia *et al.*, 2008). Additionally, these negative effects may be direct, as for example in the case of hunting (e.g. persecution of predators considered competitors) or agriculture (losses of nests or incubating birds through mechanization of practices) (see Thompson *et al.*, 2003; Woodroffe, Thirgood & Rabinowitz, 2005). Thus, conflicts between human development and wildlife conservation are common, and their resolution is a key aspect of current conservation philosophy (Conover, 2002; Woodroffe *et al.*, 2005; Macdonald & Service, 2007).

Ecotourism, fisheries and forestry exploitation are economically important anthropogenic activities that, in addition, are key sources of disturbance (Bowles, 1995; Donázar *et al.*, 2002; Arlettaz *et al.*, 2007). The creation of buffer zones into which humans are prohibited to enter, or in which certain activities are temporally restricted, may constitute a tool for minimizing the impact of the human disturbance (González *et al.*, 2007), but may be unviable in certain circumstances and even detrimental in terms of conservation if the activities that are limited are rendered economically unsustainable, and this in turn causes land use changes that are as negative for the species as the disturbance itself (or even more so). Examples of such conflicts between economic activities that are negative for a particular species, but that maintain the habitats on which these species depend are, for instance, the hunting of red grouse in Scotland, which has a negative effect on hen harrier *Circus cyaneus* conservation, but that helps to maintain its breeding habitat (Thirgood *et al.*, 2000), the timber exploitation in mature boreal forests overlapping the breeding activity of different animal species but constituting a long-term sustainable forest management and the protection of important biodiversity hotspots (Lindenmayer, Margules & Botkin, 2000; Rosenvald and Lôhmus, 2003) or burning vegetation in a sustainable way to prevent the spread of scrub and to increase the heterogeneity in grasslands for animal communities conservation (Fuhlendorf *et al.*, 2006; Spottiswoode *et al.*, 2009).

In the specific case of human activities being a source of disturbance, studies quantifying their impact on wildlife help us to understand the magnitude of the problem. It is thus particularly important to quantify the effects of disturbance not only on behaviour but also on population sustainability (via quantifying the effects of behaviour change on population parameters such as productivity or survival), such that we can measure the costs and benefits (in terms of population changes) of not limiting these activities. In this sense, it seems advisable to examine the cost-effective conservation actions before their application (Pullin *et al.*, 2004; Sutherland *et al.*, 2004). Additionally, observational studies (such as this one) aid in finding optimal solutions for harmonizing conservation and economic development when the application of buffer zones is not possible. Overall, these studies may help to decrease the tension between stakeholders by putting the conflict in an objective, traceable context.

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References

Arlettaz, R., Patthey, P., Baltic, M., Leu, T., Schaub, M., Palme, R. & Jenni-Eiermann, S. (2007). Spreading free-riding snow sports represent a novel serious threat for wildlife. *Proc. Roy. Soc. Lond. Ser. B* **274**, 1219–1224.

Arroyo, B.E. & Razin, M. (2006). Effects of human activities on bearded vulture behaviour and breeding success in the French Pyrenees. *Biol. Conserv.* **128**, 276–284.

Bautista, L.M., García, J.T., Calmaestra, R.G., Palacín, C., Martín, C.A., Morales, M.B., Bonal, R. & Viñuela, J.

(2004). Effect of weekend road traffic on the use of space by raptors. *Conserv. Biol.* **18**, 1–7.

Behera, M.D., Kushwaha, S.P.S. & Roy, P.S. (2005). Rapid assessment of biological richness in a part of Eastern Himalaya: an integrated three-tier approach. *Forest Ecol. Mgmt.* **207**, 363–384.

BirdLife International. (2008). *Aegypius monachus. IUCN Red List of Threatened Species*. Gland, Switzerland: IUCN. Available at www.iucnredlist.org.

Blumstein, D.T., Fernández-Juricic, E., Zollner, P.A. & Garity, S.C. (2005). Inter-specific variation of avian responses to human disturbance. *J. Appl. Ecol.* **42**, 943–953.

Borges, J.G., Oliveira, A.C. & Costa, M.A. (1997). A quantitative approach to cork oak forest management. *Forest Ecol. Mgmt.* **97**, 223–229.

Bortolotti, G.R., Marchant, T.A., Blas, J. & German, T. (2008). Corticosterone in feathers is a long-term, integrated measure of avian stress physiology. *Funct. Ecol.* **22**, 494–500.

Bowles, A.E. (1995). Responses of wildlife to noise. In *Wildlife and recreationists: coexistence through management and research*: 109–132. Knight, R. & Gutzwiller, K. (Eds). Washington: Island Press.

Butchart, S.H.M., Walpole, M., Collen, B., Strien, A.V., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Kapos, M.H.V., Lamarque, J.-F.,Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Morcillo, M.H., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D.,Vié, J.-C. & Watson, R. (2010). Global biodiversity: indicators of recent declines. *Science* **328**, 1164–1168.

Conover, M. (2002). *Resolving human–wildlife conflicts: the science of wildlife damage management*. Boca Raton: CRC Press.

De la Puente, J., Moreno-Opo, R. & Del Moral, J.C. (2007). *El buitre negro en España. Censo nacional 2006*. Madrid: SEO/BirdLife.

Donázar, J.A. (1993). *Los buitres ibéricos. Biología y Conservación*. Madrid: J.M. Reyero.

Donázar, J.A., Blanco, G., Hiraldo, F., Soto-Largo, E. & Oria, J. (2002). Effects of forestry and other land-use practices on the conservation of cinereous vulture. *Ecol. Appl.* **12**, 1445–1456.

Fernández-Juricic, E., Venier, M.P., Renison, D. & Blumstein, D.T. (2005). Sensitivity of wildlife to spatial patterns of recreationist behaviour: a critical assessment of minimum approaching distances and buffer areas for grassland birds. *Biol. Conserv.* **125**, 225–235.

Fuhlendorf, S.D., Harrell, W.C., Engle, D.M., Hamilton, R.G., Davis, C.A. & Leslie, D.M. (2006). Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazin. *Ecol. Appl.* **16**, 1706–1716.

Gill, J.A. (2005). Approaches to measuring the effects of human disturbance on birds. *Ibis* **149**, 9–14.

Gill, J.A., Norris, K. & Sutherland, W.J. (2001). Why behavioural responses may not reflect the population consequences of human disturbance. *Biol. Conserv.* **97**, 265–268.

González, L.M., Arroyo, B.E., Margalida, A., Oria, J. & Sánchez, R. (2006). Effect of human activities on the behaviour of breeding Spanish imperial eagles (*Aquila adalberti*): management implications for the conservation of a threatened species. *Anim. Conserv.* **9**, 85–93.

González, L.M., Arroyo, B.E., Margalida, A., Oria, J. & Sánchez, R. (2007). Can buffer zones have detrimental effects on conservation of Spanish imperial eagle *Aquila adalberti*? *Anim. Conserv.* **10**, 295–296.

Guil, F. & Moreno-Opo, R.(Coords). (2008). *Guía práctica para la elaboración de planes de gestión Natura 2000 en fincas y montes*. Madrid: Obra Social Caja Madrid-Fundación CBD-Habitat.

Hiraldo, F. (1983). Breeding biology of the cinereous vulture. In *Vulture biology and management*: 197–213. Wilbur, S.R. & Jackson, J.A. (Eds). Berkeley: University of California Press.

Holmes, N., Giese, M. & Kriwoken, L.K. (2005). Testing the minimum approach distance guidelines for incubating royal penguins *Eudyptes schlegeli*. *Biol. Conserv.* **126**, 339–350.

Langston, R.H.W., Liley, D., Murison, G., Woodfield, E. & Clarke, R.T. (2007). What effects do walkers and dogs have on the distribution and productivity of breeding European Nightjar *Caprimulgus europaeus*? *Ibis* **149**, 27–36.

Lindenmayer, D.B, Margules, C.R. & Botkin, D.B. (2000). Indicators of biodiversity for ecologically sustainable forest management. *Conserv. Biol.* **14**, 941–950.

Macdonald, D. & Service, K. (2007). *Hot topics in conservation*. Oxford: Blackwell Publishing.

Margalida, A., Donázar, J.A., Carrete, M. & Sánchez-Zapata, J.A. (2010). Sanitary versus environmental policies: fitting together two pieces of the puzzle of European vulture conservation. *J. Appl. Ecol.* **47**, 931–935.

Marton-Lefèvre, J. (2010). Biodiversity is our life. *Science* **313**, 1051.

Meretsky, V., Snyder, N., Beissinger, S. & Clendenen, D.J. (2000). Demography of the California condor: implications for reestablishment. *Conserv. Biol.* **14**, 957–967.

Ministerio de Medio Ambiente, Medio Rural y Marino (2009). *Estadísticas agrarias*. Available at http://www.mapa.es/es/estadistica/pags/preciospercibidos/metodologia/metodologia.html (accessed 10 September 2010).

Moreno-Opo, R. & Arredondo, A. (2007). Descorche y buitre negro. In *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*: 296–306. Moreno-Opo, R. & Guil, F. (Eds). Madrid: Ministerio de Medio Ambiente.

Oro, D., Margalida, A., Carrete, M., Heredia, R. & Donázar, J.A. (2008). Testing the goodness of supplementary feeding to enhance population viability in an endangered vulture. *PLoS One* **3**, e4084.

Pereira, H. (2007). *Cork. Biology, production and uses*. Oxford: Elsevier Ltd.

Preisler, H.K., Ager, A. & Wisdom, M.J. (2006). Statistical methods for analysing responses of wildlife to human disturbance. *J. Appl. Ecol.* **43**, 164–172.

Pullin, A.S., Knight, T.M., Stone, D.A. & Charman, K. (2004). Do conservation managers use scientific evidence to support their decision-making? *Biol. Conserv.* **119**, 245–252.

Richardson, C.T. & Miller, C.K. (1997). Recommendations for protecting raptors from human disturbance: a review. *Wildl. Soc. Bull.* **25**, 634–638.

Rosenvald, R. & Lõhmus, A. (2003). Nesting of the black stork *Ciconia nigra* and white-tailed eagle *Haliaeetus albicilla* in relation to forest management. *Forest Ecol. Mgmt.* **185**, 217–223.

Sergio, F., Blas, J., Forero, M., Fernandez, N., Donázar, J.A. & Hiraldo, F. (2005). Preservation of wide-ranging top predators by site-protection: black and red kites in Doñana National Park. *Biol. Conserv.* **125**, 11–21.

Speziale, K.L., Lambertucci, S.A. & Olsson, O. (2008). Disturbance from roads negatively affects Andean condor habitat use. *Biol. Conserv.* **141**, 1765–1772.

Spottiswoode, C.N., Wondafrash, M., Gabremichael, M.N., Dellelegn, Y., Mwangi, M.A.K., Collar, N.J. & Dolman, P.M. (2009). Rangeland degradation is poised to cause Africa's first recorded avian extinction. *Anim. Conserv.* **12**, 249–257.

Steidl, R.J. & Anthony, R.G. (2000). Experimental effects of human activity on breeding bald eagles. *Ecol. Appl.* **10**, 258–268.

Sutherland, W.J., Pullin, A.S., Dolman, P.M. & Knight, T.M. (2004). The need for evidence-based conservation. *Trends Ecol. Evol.* **19**, 305–308.

Thirgood, S.J., Redpath, S.M., Newton, I. & Hudson, P.J. (2000). Raptors and grouse: conservation conflicts and management solutions. *Conserv. Biol.* **14**, 95–104.

Thompson, D.B.A., MacDonald, A.J., Marsden, J.H. & Galbraith, C.A. (1995). Upland heather moorland in Great Britain: a review of international importance, vegetation change and some objectives for nature conservation. *Biol. Conserv.* **71**, 163–178.

Thompson, D.B.A., Redpath, S.M., Fielding, A.H., Marquiss, M. & Galbraith, C.A. (Eds). (2003). *Birds of prey in a changing environment*. Edinburgh: The Stationery Office.

Whitfield, D.P., Ruddock, M. & Bullman, R. (2008). Expert opinion as a tool for quantifying bird tolerance to human disturbance. *Biol. Conserv.* **141**, 2708–2717.

Woodroffe, R., Thirgood, S.J. & Rabinowitz, A. (Eds). (2005). *People and wildlife: conflict or co-existance?* Cambridge: Cambridge University Press.

WWF. (2006). *Cork screwed? Environmental and economic impacts of the cork stoppers market. A WWF Report*. WWF/MEDPO. Available at http://assets.wwfes.panda.org/downloads/corcho_amenazado_ingles.pdf (accessed 10 September 2010).

Young, J., Watt, A., Nowicki, P., Alard, D., Clitherow, J., Henle, K., Johnson, R., Laczko, E., McCracken, D., Matouch, S., Niemela, J. & Richards, C. (2005). Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. *Biodivers. Conserv.* **14**, 1641–1671.

Zuberogoitia, I., Zabala, J., Martínez, J.A., Martínez, J.E. & Azkona, A. (2008). Effect of human activities on Egyptian vulture breeding success. *Anim. Conserv.* **11**, 313–320.

CAPÍTULO-CHAPTER
6

Linking sanitary and ecological requirements in the management of avian scavengers: effectiveness of fencing against mammals in supplementary feeding sites

by

Rubén Moreno-Opo, Antoni Margalida, Francisco García, Ángel Arredondo, Carlos Rodríguez & Luís Mariano González (2012)

Biodiversity and Conservation **21: 1673-1685**

Uniendo requisitos sanitarios y ecológicos en la conservación de aves necrófagas: eficacia de vallados ante mamíferos en puntos de alimentación suplementaria

La compatibilización de distintas políticas sectoriales resulta de gran importancia para asegurar el cumplimiento de los objetivos de conservación de la biodiversidad. Este el caso de la aplicación de medidas de control sanitario a la gestión de los subproductos animales no destinados a consumo humano y sus efectos negativos en la dinámica poblacional y ecología comportamental de algunas especies de aves necrófagas. Por ello, es preciso adoptar medidas que permitan la alimentación de estas especies y que paralelamente garanticen la ausencia de riesgos de transmisión de enfermedades. El presente estudio tiene como objetivo la mejora en el desarrollo técnico de una de las herramientas más utilizadas por gestores y conservacionistas para la conservación de aves necrófagas: la alimentación suplementaria en comederos vallados. De acuerdo con la normativa vigente que obliga en determinadas circunstancias el cercado de estos enclaves, se evaluó la permeabilidad a distintas especies de cuatro modelos de cercados, para conocer cuáles de ellos satisfacen los requisitos sanitarios de impedir el acceso de especies necrófagas no objetivo de la alimentación vectores potenciales de determinadas enfermedades transmisibles. Los ensayos de aportes en cercados se compararon respecto a aportes aleatorios (control) en emplazamientos no cercados. Los resultados muestran cómo dos de los modelos (*red alta* y *red baja*) impiden la entrada de mamíferos necrófagos facultativos durante un período de más de dos meses y de entre siete y ocho aportes, tras la realización del primero de ellos en el mismo punto. Las aves necrófagas accedieron al interior para consumir los aportes independientemente del tipo de cercado. Como consecuencia de estos resultados, se plantean recomendaciones de gestión para la alimentación suplementaria de aves amenazadas y de manejo de los cercados ensayados.

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ORIGINAL PAPER

Linking sanitary and ecological requirements in the management of avian scavengers: effectiveness of fencing against mammals in supplementary feeding sites

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Abstract In order to ensure that the objectives behind the conservation of biodiversity are fulfilled it is essential that policies of all stakeholders are compatible. This is the case of the application of sanitary measures for the management of animal by-products and the negative effects that such restrictions had on the population dynamics and behavioural ecology of the avian scavengers' guild. Thus, measures that allow these species to feed and that reduce risks of disease transmission must be put into practice. This study aims to improve the technical implementation of one of the commonest tools employed in the conservation of avian scavengers: supplementary feeding stations. We evaluated the permeability of three types of fences in experimental feeding stations to determine which of the models prevent non-target species from accessing the food provided. We compared

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results from fenced-off feeding stations with those from random points in unfenced-off sites. The results showed that two of the models (*high* and *low nets*) were the most effective avoiding facultative mammal scavengers from gaining access into the exclosure for over two months and for 7–8 inputs of food. Avian scavengers were able to access food regardless of the type of exclosure, which did not determine the abundance of birds or the species present. The carrion consumption by non-target species can be reduced by affordable and sustainable means. We suggest proposals to optimize the management of supplementary feeding stations for threatened avian scavengers and for the design of fenced exclosures.

Keywords Animal by-products · Health legislation · Scavenger · Spain · Feeding programs · Vulture conservation · Vulture restaurant

Introduction

Avian scavengers provide important ecosystem services in balancing trophic chains by completing the processing cycle and assimilation of biomass of dead animals (DeVault et al. 2003; Şekercioğlu et al. 2004). Scavenger species have evolved behavioural and morphological mechanisms that enable them to optimize the way in which they exploit carrion (König 1983; Donázar 1993; Hertel 1994). The traditional presence of dead animals in the wild has led to the development of complex ecological relationships between soils, vegetation and primary and secondary consumers (DeVault et al. 2003; Whelan et al. 2008). Yet, various scientific sources have revealed that the interaction among wild and domestic animals and humans can lead to the spread of certain diseases (Caley and Hone 2004), which may negatively affect human and animal health and increase costs in farming practices, conservation actions and sanitary measures (Daszak et al. 2000; Horan and Wolf 2005; Gortázar et al. 2008). Furthermore, these interactions have been proposed as the cause of spreading of emerging diseases that are potentially threatening for the world's population health (Schiermeier 2001; Kilpatrick et al. 2006). In order to avoid sanitary risks it is vital that the management techniques of animal carcasses be as appropriate as possible (Gortázar et al. 2007; Maichak et al. 2009).

As a means of avoiding risks to human and animal health, sanitary authorities in the European Union (EU) have set up in recent years initiatives aimed at regulating the use and exploitation of animal by-products that could become potential vectors of transmittable diseases (Donázar et al. 2009b). EU legislation has been issued in the last decade whose intention is to control the management of animal by-products not intended for human consumption (Regulation CE 1774/2002); from the onset it was perceived that such controls were likely to affect the occurrence and availability of food resources as well as to the foraging patterns of avian scavengers (Tella 2001). In particular, such sanitary legislation only exempted the compulsory carcasses destruction in certain cases for allowing their transfer to only enclosed avian scavenger feeding stations whose fences would prevent access by mammals and thus the propagation of disease.

The management of food resources exclusively via specific enclosed feeding points has been shown to be insufficient in terms of availability for the Iberian avian scavenger populations (García de Francisco and Moreno-Opo 2009). Moreover, since the obligation of providing carrion in fenced feeding sites alterations in foraging patterns, dispersion and even behaviour have been detected (Deygout et al. 2009; Margalida et al. 2010; Zuberogoitia et al. 2010), which have led to changes in species' population dynamics, the appearance of the negative effects associated with predictable availability of food, and negative social and economic impacts on the interests of rural communities (Carrete et al. 2006; Robb et al. 2008; Margalida et al. 2011b). In light of this situation, as of 2011 the EU have loosened restrictions in certain circumstances and new aspects related to the natural patterns of feeding behaviour in avian scavengers have been incorporated into the legislation (Council of Europe 2009; European Commission 2011). As a result, some carcasses may be left in the wild without prior collection and without the obligation of placing in a fenced-off area.

After this modification of the sanitary legislation, it is expected that supplementary feeding programs could allow a higher availability and wider occurrence of carcasses, improving the food quality, more consistent with the ecological requirements of avian scavengers (Margalida et al. 2010). Similarly, no obligation of fencing all the supplementary feeding points is likely to reduce the economic costs to administrations and managers as well as the negative effects that food concentration at a few points implies in the behavioural ecology of avian scavengers (Carrete et al. 2006; Francisco de García and Moreno-Opo 2009; Cortés-Avizanda et al. 2010; Margalida et al. 2010). On the other hand, it may be advisable in certain circumstances the use of technical systems that prevent the access of potential disease vectors to the provided carcasses. In these cases, and in order to provide a quick and healthy consumption of by-products by avian scavengers, it would be necessary to have effective, manageable and affordable designs to limit access to the carcass only to the target species.

This study aims to contribute to making sanitary requirements compatible with ecological needs in the supplementary feeding of threatened avian scavengers. It tries to implement the new EU legislation on the management of animal by-products, proposing best practical fencing models and meeting certain positive requirements for all the involved stakeholders: threatened scavengers feed on a sustainable way, health risks are avoided and economic costs are reduced in relation to conventional feeding stations. The specific objectives of the study are thus as follows: 1) to evaluate the permeability of different types of exclosures and the ability of different scavenger species to enter fenced-off areas and 2) to promote cheap, mobile and easily manageable models of exclosures that not only prevent facultative mammalian scavengers from entering feeding stations, but also replicate as much as possible the natural appearance of carrion for avian scavenger species.

Materials and methods

Studied area and species

Two supplementary feeding areas were chosen in Castilla-La Mancha in central Spain (Fig. 1): in Los Yébenes (Montes de Toledo) food was provided in exclosures (treatment), whilst four sites at Almodóvar del Campo and Fuencaliente (Sierra Morena) were treated as controls.

Similar scavenger communities in terms of composition and relative abundance exist in both areas (Del Moral and Martí 2003). The following avian scavenger guild were studied: Eurasian griffon vulture *Gyps fulvus* (<20 breeding pairs in a radius of 10 km around the feeding stations, although there are also many non-breeding present in both treatment and control areas throughout the year); cinereous vulture *Aegypius monachus* (the feeding stations are less than 5 km from the two largest colonies of this species in Castilla-La Mancha: 165 pairs in the treatment area and 129 in the control area; De la Puente et al.

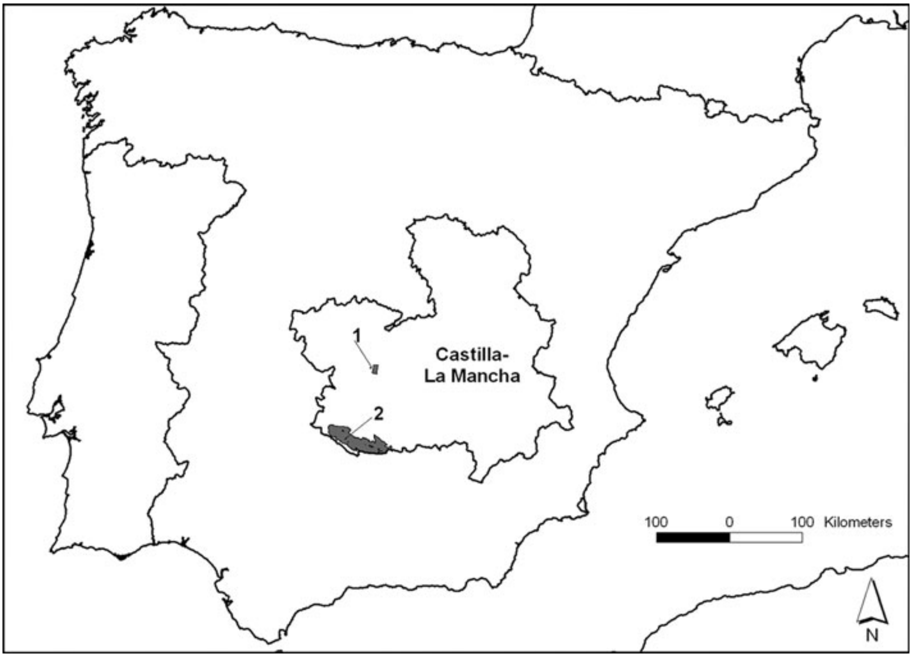

Fig. 1 Location of study sites in Castilla-La Mancha, central Spain. Study sites were in Montes de Toledo (1) and in Sierra Morena (2)

2007); common raven *Corvus corax* (2–5 pairs in a radius of 10 km from the feeding points and presence of flocks in winter in treatment and control areas); and the Spanish imperial eagle *Aquila adalberti* (3–5 pairs in a radius of 10 km of the feeding station and presence of non-breeders in treatment and control areas). The facultative mammalian scavengers present are red fox *Vulpes vulpes* and wild boar *Sus scrofa*. The relative abundance of these mammals in both the treatment and control areas were not calculated, although they were considered to have similar values given the similarities between both areas in terms of their socio–economic models, habitats, hunting statistics and the potential abundance of available resources (Virgós 2002; Vicente et al. 2007; Mangas and Rodríguez-Estival 2010). Dogs *Canis familiaris* were also considered during this study given the presence at the end of the hunting season of animals escaped from hunters (R. Moreno-Opo, A. Arredondo and F. García, unpublished data). The presence of red deer *Cervus elaphus* was also taken into account given that, despite not being a scavenger, it is attracted to carrion and as such could become a vector of transmittable diseases (Gortázar et al. 2008).

Field work and variables

Exclosure models were chosen from livestock and other properties mobile protection systems, among the best available options. Then, three models were selected according to their compliance of the prior requisites of the study: (a) low economic cost in comparison with conventional feeding stations—that typically have fences of 2 m high with overhangs, fixed posts and cover c. 1 ha (i.e. Consejería de Medio Ambiente 2006), (b) a surface area of at least 0.5 ha and (c) can be easily set up and removed. The following types of exclosures were tested:

– *Ribbons*: electrified ribbons (5–8 volts) and metal posts enclosing a 75 × 75 m; four parallel lines of white polythene ribbons with 20 mm steel conductors attached by means of plastic insulators to posts; as well, a galvanized steel wire (diameter 0.4 mm) placed 20 cm from the ground and 20 cm in front of the tapes; gate kit and Viper S250 solar-powered fence charger equipped with rechargeable battery (Fig. 2A).
– *Low net*: a 90 cm high electrified mesh fence (5–8 volts), enclosing an area of 75 × 75 m; orange polythene net woven with three 0.25 mm wire conductors; 17 cm^2 mesh sized; plastic posts and Viper S250 solar-powered fence charger equipped with rechargeable battery (Fig. 2B).
– *High net*: a 170 cm high electrified mesh fence (5–8 volts), enclosing an area of 75 × 75 m; orange polythene mesh woven with three 0.25 mm wire conductors; 17 cm^2 mesh sized; plastic posts and Viper S250 solar-powered fence charger equipped with rechargeable battery (Fig. 2C).

The conventional feeding stations authorized by official bodies (Consejería de Medio Ambiente 2006) were not studied since they do not fulfil the previously established requisites regarding cost, mobility and size.

The three tested models were set up and activated for a period of 4 months (Table 1), by using one exclosure of each model at one site. Between April 2010 to April 2011 carrion was supplied every fortnight to control sites (without fencing; $n = 4$ sites and $n = 40$ inputs of food) and treatment areas ($n = 46$ whole inputs of food for all the exclosures, see Table 1 for details) to test their respective permeability to scavenger species. Inputs of food consisted of remains of red deer and wild boar in both control and treatment areas ($n = 66$), as well as goat *Capra hircus* ($n = 3$) and rabbit *Oryctolagus cuniculus* var. dom. ($n = 17$) carcasses. The average quantity of carrion supplied each time was 75.0 ± 37.8 kg (range 40–180) for *ribbons*, 74.4 ± 48.7 kg (range 30–200) for *low net*, 55.9 ± 25.6 kg (range 25–120) for *high net* and 85.5 ± 147.8 kg (range 5–450) for the *controls*.

Two automatic photo-cameras were placed at 5 m from the carrion to detect and identify which species were feeding on and thus which species were able to penetrate into the feeding station. Specifically, Scoutguard SG550 (HCO Outdoor Products, Norcross, USA) cameras were used, equipped with movement and infrared sensors for night-time use, and programmed in phases of three photos with a delay of one minute between phases. The cameras were checked and images downloaded every fortnight. After images were analysed, the following variables were noted: type of exclosure, date on which food was supplied, quantity of biomass supplied (these three variables not obtained from the photo analysis), maximum number of each feeding species present simultaneously, and number of days elapsing between the food supply and the arrival of each feeding species.

Statistical analyses

Differences between the different types of exclosure were analyzed in terms of the number of days (from the beginning of the food supply) until each of the study species was detected for the first time and in relation to the number of individuals accessing the food. As a result, two response variables were selected: 1) the number of days that the scavengers took to come to feed since the input of the food, and 2) the maximum number of individuals of

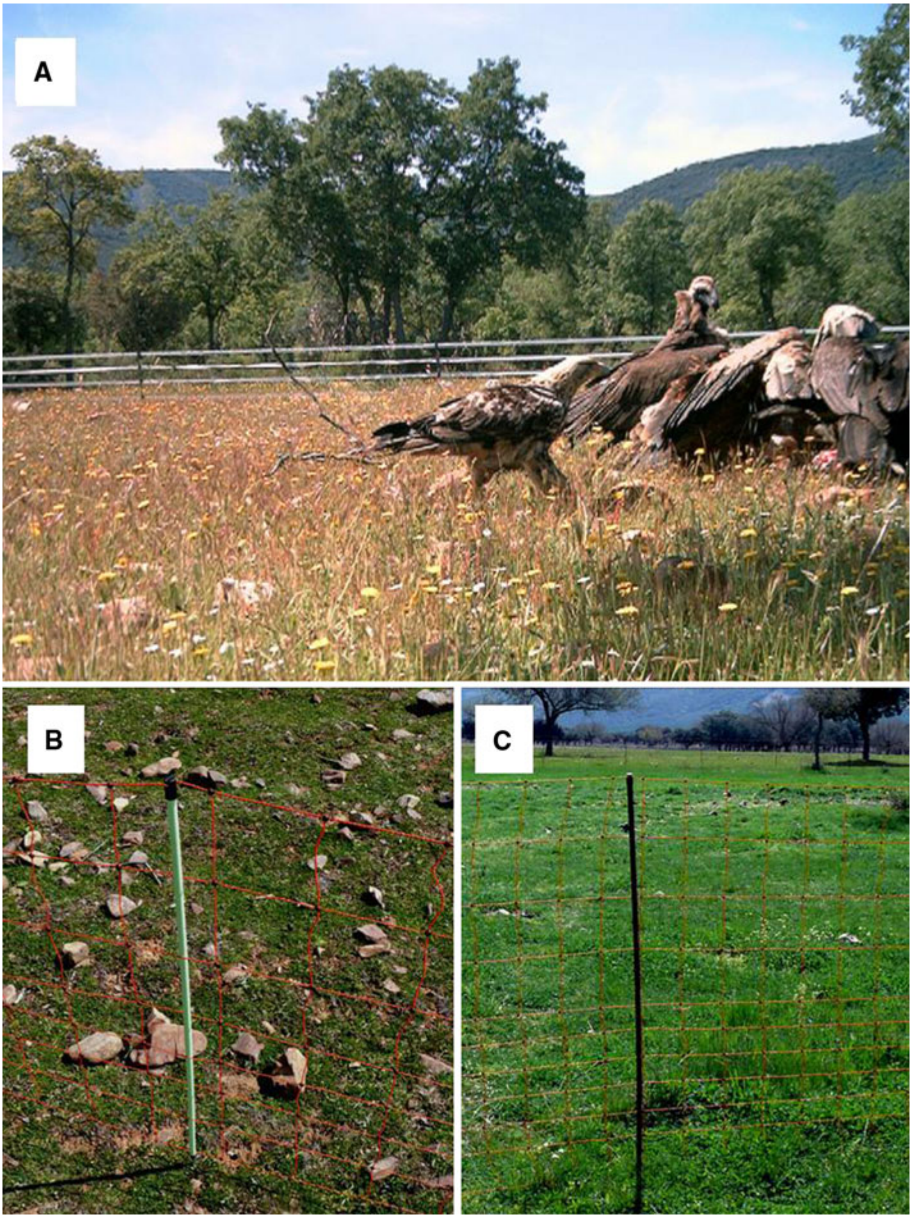

Fig. 2 Examples of the fence types used for exclosures: **A** *Ribbons*—this can be seen behind the foraging birds (*Aquila adalberti*, *Aegypius monachus* and *Gyps fulvus*); **B** *Low net*; **C** *High net* (see text for detailed descriptions of each fence type)

each species counted at each input of food, as the maximum simultaneous number registered in the photos.

Both of these variables were analyzed in relation to the following independent variables: 1) exclosure type, 2) the total number of days that the supplementary feeding site

Table 1 Summary of carrion inputs used during the study by treatment type. Inputs (n) = total number of carrion inputs; Days = number of days carrion was available; Date = the dates on which the treatments remained active; Biomass (kg) = the total mass of carrion provided in kilograms

Fence models	Inputs (n)	Days	Date	Biomass (kg)
Ribbons	17	125	6th May to 8th Sept 2010	1,275
Low net	17	142	19th April to 7th Sept 2010	1,265
High net	11	110	14th Dec 2010 to 5th April 2011	615
Control (no fenced)	40	202	24th April 2010 to 31st Jan 2011	3,420

was active (Table 1) and, 3) the total biomass (kg) of food supplied in each exclosure type (Table 1).

The analysis consisted of a General Linear Model (GLM) with a Poisson-type family of errors, using as fixed factors each of the three independent variables and with a confidence interval of 95 %. Only the significant results and those that approach significance are presented. The analyses were carried out using the software Statistica 6.1 (StatSoft 2002).

Results

In total, 6,655 kg of carrion on 86 different inputs were supplied, and a total of 2,561 griffon vultures, 617 cinereous vultures, 127 ravens, nine Spanish imperial eagles, 45 foxes, 33 wild boars, nine dogs and 24 red deer could be counted (obtained by summing the maximum number of simultaneous individuals present at each input of food).

Only significant values are presented due to the great amount of models performed ($n = 16$). In terms of the accessibility into the different exclosures, there were no significant differences in the number of days it took for the birds to accede to the carrion. On the other hand, the time taken by foxes to get to the carrion after its supply was greatest when the feeding site had a *low net* and *high net* fence ($F = 9.04$, df = 4, $P = 0.001$). Thus, these two models resulted more successful in delaying the entrance of the fox for a greater number of inputs and for a longer period (Table 2, Fig. 3); wild boar, dogs and red deer, on the other hand, did not get into the exclosures and were only recorded at control

Table 2 Time taken by different mammal species to access the supplied carrion, expressed as both the ordinal number of carrion inputs and the number of days prior to entrance of each species since activation of the feeding site. For the controls, each input was considered as independent for the four different study sites so mean values (±S.D.) are shown for the day of first recorded entrance to feed

	Fox		Wild boar		Dog		Red deer	
Fence models	Input	Day	Input	Day	Input	Day	Input	Day
Ribbons	1st	0	No access	No access	No access	No access	No access	No access
Low net	7th	62	No access	No access	No access	No access	No access	No access
High net	8th	64	No access	No access	No access	No access	No access	No access
Controls (no fenced)	1st	3.1 ± 4.6	2nd	7.2 ± 9.8	1st	9.0 ± 1.4	1st	2.3 ± 0.5

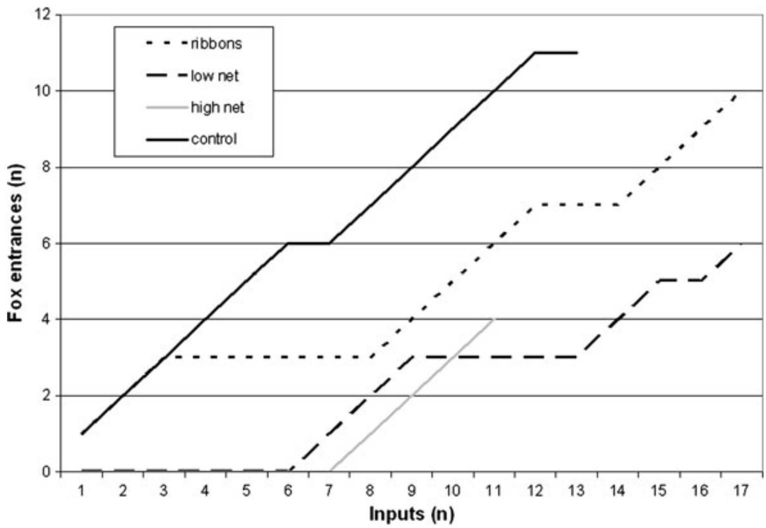

Fig. 3 Number of fox *Vulpes vulpes* entrances to the carrion provided within each site using different types of fencing in relation to the number of carrion inputs to each site

sites without fences. Time needed for foxes to access the food was less when the amount of biomass was greater ($F = 4.66$, df $= 1$, $P = 0.037$).

In relation to the number of individuals accessing to the carrions, fewer foxes were recorded in the exclosures, mainly in the *low net* and *high net* models ($F = 2.36$, df $= 4$, $P = 0.058$) than at the unfenced (control) sites. Wild boar became more abundant when the feeding point was active longer (in the case of non-fenced points, $F = 6.15$, df $= 1$, $P = 0.015$). Only in the case of the cinereous vulture the number of individuals resulted related to the type of fence where carrion was provided ($F = 5.06$, df $= 4$, $P = 0.011$, Fig. 4); this species was more abundant at control feeding sites without perimeter fences. An increase in the number of days a feeding station was operative ensured that the number of cinereous vultures, ravens and Spanish imperial eagles also increased ($F = 19.58$, df $= 1$, $P = 0.001$; $F = 6.05$, df $= 1$, $P = 0.016$; $F = 3.82$, df $= 1$, $P = 0.054$, respectively). Likewise, the supply of a greater quantity of biomass involved that more griffon and cinereous vultures were present at the feeding site ($F = 179.00$, df $= 1$, $P = 0.001$; $F = 49.73$, df $= 1$, $P = 0.001$, respectively).

Discussion

Feeding of targeted and non-targeted species

These results constitute the first obtained from an experiment aimed at optimizing the management of supplementary feeding sites for scavenger species, and throw light on the patterns of access to carrion protected by different types of perimeter exclosures by a scavenger guild. To date only few studies have been published on this subject (Cortés-Avizanda et al. 2010) and so we are unable to contrast our results with others from other

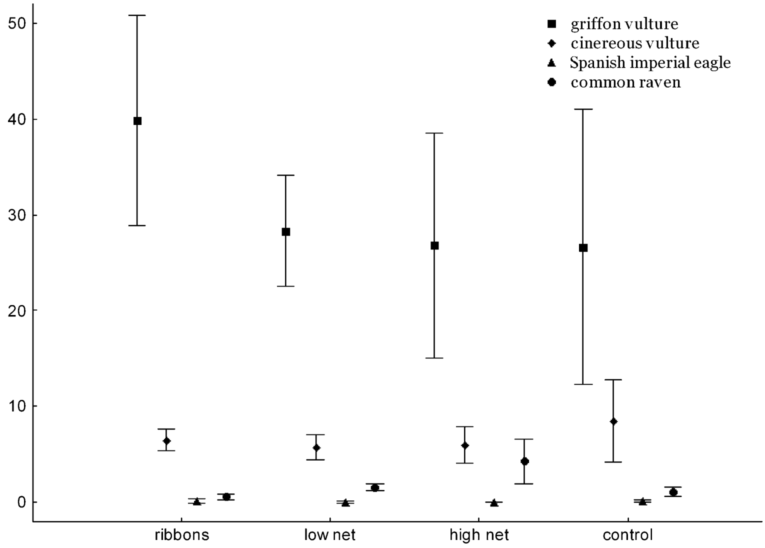

Fig. 4 Mean number of individuals of each scavenger bird species (±95 % confidence interval) recorded in each exclosure type

geographical areas or other types of exclosures. The reasons for this lack of previous global comparative studies may be due to the fact that only within the EU the recent legal sanitary framework has lead to avian scavengers to several conservation problems associated with changes in the availability of food resources (Donázar et al. 2009a; Margalida et al. 2012). In this sense, given the serious conservation problems affecting vulture populations in the South Asian region in recent years (Oaks et al. 2004; Shultz et al. 2004; Gilbert et al. 2007) and the incipient detected threat for African vulture populations due to loss of food quality (Naidoo et al. 2009; Virani et al. 2011), the information contained in our study could be of interest for managing of supplementary feeding sites as an effective conservation tool (Markandya et al. 2008).

Our results showed that no avian scavenger avoided entering the tested exclosures. This is highly relevant for the development of management protocols for feeding stations, despite considerations about their suitability and usefulness (Piper 2006; Robb et al. 2008). Although very few studies have tackled the subject of the availability of food resources and their effects on population dynamics of scavenger species (Colomer et al. 2011; Margalida et al. 2011a), different public administrations have already set up supplementary feeding programmes through conventional feeding stations in order to palliate the problems regarding carrion availability (García de Francisco and Moreno-Opo 2009). Such conventional structures of feeding station comply with sanitary regulations and prevent non-target mammal species from entering (if the appropriate maintenance is carried out). These models have had positive effects and have helped mitigate the shortage of carrion and its lack of quality that has occurred in recent years (Gilbert et al. 2007; Hernández and Margalida 2008; Oro et al. 2008; García de Francisco and Moreno-Opo 2009). In addition, for threatened species such as the Egyptian vulture *Neophron percnopterus* these feeding stations have performed a fundamental role in the exchange of information among birds and the substitution of individuals in breeding pairs via the establishment of communal roosting sites (Donázar et al. 1996; Benítez et al. 2009) and probably by reducing the high mortality rates related with the illegal use of poisoning baits (Hernández and Margalida 2009).

Management of supplementary feeding sites

The two models that best prevent facultative scavenger mammals from entering the exclosures were the *low net* and *high net* (Figs. 3, 4) and are the most recommendable structures for fulfilling the objectives proposed at the beginning of this study. Furthermore, both types are highly manageable and easy to install, remove and move to another site given that there is no need to fix posts permanently or to set up a system of insulators. The cost of each exclosure (0.5 ha) is 900 €, as opposed to ~9,000 € in the case of fixed conventional feeding stations.

In terms of the efficiency, we recommend that exclosures are checked and maintained for a suitable working. Moreover, regarding the permanence, exclosures may be kept at the same site for a maximum of around 2 months so as to prevent opportunist mammals such as red foxes learning how to find a way in. The presence of abundant food resources at the same site for an over-long period of time will also increase its attraction vis-a-vis other foraging areas and thus modify certain ecological processes and relationships (Cortés-Avizanda et al. 2009; Deygout et al. 2009; Donázar et al. 2009b). Once the two-month period finishes, it is advisable to then move the feeding point elsewhere, preferably as far away as possible. In this way the temporal and spatial unpredictability of the natural appearance of carrion is imitated, sanitary conditions are guaranteed and the management of supplementary feeding of avian scavengers can be optimized (Olea and Mateo-Tomás 2009).

The suggested models could be highly appropriate for sectors that are directly involved in the management of livestock and game animal by-products. Farmers and hunting managers could perceive as positive these types of installations and hence may encourage their implementation. In this way, the territorial scope of these models could be broadened. This active participation may help lessen any reticence regarding the presence of avian scavengers and deter the use of certain types of illegal predator control methods (Koenig 2006; Hernández and Margalida 2008, 2009; Margalida et al. 2011b), since these carrion provided in the proposed exclosures, unlike those occurring naturally in the field (Cortés-Avizanda et al. 2010), are not accesible to opportunistic predators and do not favor the growth of their populations. The fact that 1) the cost of purchasing and installing this type of feeding stations is much less than that of conventional exclosures, that 2) the possibility of their use in periods that are beneficial for the management and exploitation in the estates (hunting, culling for sheeps and goats, and lambing seasons), and that 3) it is not necessary to intend permanently a large land surface of the settlement of a feeding station (up to 1 ha) are three reasons that could forward that the use of these types of feeding stations are practical; generalizable and appropriate to many rural scenarios (Piper 2006; Margalida et al. 2010).

Regardless questions arising from management issues, it is important to monitor the supplementary feeding measures arranged (Buckland et al. 2005; Oro et al. 2008) with regard to the presence of scavenger species, their relative abundance, their feeding selection patterns and the detection of possible conservation problems as human disturbances or the lack of available food in the surrounding areas (Cortés-Avizanda et al. 2010; Moreno-Opo et al. 2010). This could be essential if we aim to evaluate the efficacy of the implemented measures and, if necessary, modify certain aspects of the management protocols in order to increase the positive effects on target species (McCarthy and Possingham 2007).

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References

Benítez JR, Cortés-Avizanda A, Ávila E, García R (2009) Effects of the creation of a supplementary feeding station for the conservation of Egyptian vulture (*Neophron percnopterus*) population in Andalucía (southern Spain). In: Donázar JA, Margalida A, Campión D (eds) Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology, Munibe 29. Sociedad de Ciencias Aranzadi San Sebastián, Spain, pp 276–291

Buckland ST, Magurran AE, Green RE, Fewater RM (2005) Monitoring change in biodiversity through composite indices. Philos Trans R Soc B Sci 360:243–254

Caley P, Hone H (2004) Disease transmission between and within species and the implications in disease control. J Appl Ecol 41:94–104

Carrete M, Donázar JA, Margalida A (2006) Density-dependent productivity depression in Pyrenean bearded vultures: implications for conservation. Ecol Appl 16:1674–1682

Colomer MA, Margalida A, Sanuy D, Pérez-Jiménez M (2011) A bio-inspired computing model as a new tool for modeling ecosystems: the avian scavengers as a case study. Ecol Model 222:33–47

Consejería de Medio Ambiente (2006) Decreto 108/2006, por el que se regula la alimentación de aves rapaces necrófagas con cadáveres y restos de animales de especies de ganadería o cinegéticas y se crea una red de muladares den el ámbito territorial de Castilla-La Mancha. D O Castilla-La Mancha 202:19924

Cortés-Avizanda A, Carrete M, Serrano D, Donázar JA (2009) Carcasses increase the probability of predation of ground-nesting birds: a caveat regarding the conservation value of vulture restaurants. Anim Conserv 12:85–88

Cortés-Avizanda A, Carrete M, Donázar JA (2010) Managing supplementary feeding for avian scavengers: guidelines for optimal design using ecological criteria. Biol Conserv 143:1707–1715

Council of Europe (2009) Regulation CE 1069/2009 of the Parliament and Council of Europe of 21st October. Off J Eur Union 300:1–33

Daszak P, Cunningham AA, Hyatt AD (2000) Emerging infectious diseases of wildlife-threats to biodiversity and human health. Science 287:443–449

De la Puente J, Moreno-Opo R, Del Moral JC (2007) El buitre negro en España. Censo nacional 2006. SEO/BirdLife, Madrid

Del Moral JC, Martí R (2003) Atlas de las aves reproductoras de España. SEO/BirdLife-Ministerio de Medio Ambiente, Madrid

DeVault TL, Rhodes OE, Shivik JA (2003) Scavenging by vertebrates: behavioural, ecological and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. Oikos 102:225–234

Deygout C, Gault A, Sarrazin F, Bessa-Gomes C (2009) Modelling the impact of feeding stations on vulture scavenging service efficiency. Ecol Model 220:1826–1835

Donázar JA (1993) Los buitres ibéricos. In: Reyero JM (ed) Madrid

Donázar JA, Ceballos O, Tella JL (1996) Dormideros comunales de alimoche *Neophron percnopterus* en el valle del Ebro: su importancia para la conservación de la especie. An Orn Navarra 2:19–31

Donázar JA, Margalida A, Campión D (2009a) Vultures feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology, Munibe 29. Sociedad de Ciencias Aranzadi San Sebastián, Spain

Donázar JA, Margalida A, Carrete M, Sánchez-Zapata JA (2009b) Too sanitary for vultures. Science 326:664

European Commission (2011) Regulation CE 142/2011, of 25th February, implementing Regulation CE 1069/2009 of the European Parliament. Off J Eur Union 54:1–254

García de Francisco JM, Moreno-Opo R (2009) Livestock carcass management today: is there enough flexibility to deal with new conservation strategies? In: Donázar JA, Margalida A, Campión D (eds) Vultures feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology, Munibe 29. Sociedad de Ciencias Aranzadi San Sebastián, Spain, pp 492–509

Gilbert M, Watson RT, Ahmed S, Asim M, Johnson JA (2007) Vulture restaurants and their role in reducing diclofenac exposure in Asian vultures. Bird Conserv Int 17:63–77

Gortázar C, Ferroglio E, Höfle U, Frölich K, Vicente J (2007) Diseases shared between wildlife and livestock: a European perspective. Eur J Wildl Res 53:241–256

Gortázar C, Torres MJ, Vicente J, Acevedo P, Reglero M, De la Fuente J, Negro JJ, Aznar J (2008) Bovine tuberculosis in Doñana biosphere reserve: the role of wild ungulates as disease reservoirs in the last Iberian Lynx strongholds. PLoS ONE 3:e2776

Hernández M, Margalida A (2008) Pesticide abuse in Europe: effects on the Cinereous vulture (*Aegypius monachus*) population in Spain. Ecotoxicology 17:264–272

Hernández M, Margalida A (2009) Poison-related mortality effects in the endangered Egyptian vulture (*Neophron percnopterus*) population in Spain. Eur J Wildl Res 55:415–423

Hertel F (1994) Diversity in body size and feeding morphology within pasta and present vulture assemblages. Ecology 75:1074–1084

Horan RDD, Wolf CAA (2005) The economics of managing infectious wildlife disease. Am J Agric Econ 87:537–551

Kilpatrick AM, Chmura AA, Gibbons DW, Fleischer RC, Marra PP, Daszak P (2006) Predicting the global spread of H5N1 avian influenza. Proc Natl Acad Sci U S A 103:19368–19373

Koenig R (2006) Vulture research soars as the scavengers' numbers decline. Science 312:1591–1592

König C (1983) Interspecific and intraspecific competition for food among old world vultures. In: Vulture biology and management. University of California Press, Berkeley, pp 153–171

Maichak EJ, Scurlock BM, Rogerson JD, Meadows LL, Barbknecht AE, Edwards WH, Cross PC (2009) Effects of management, behaviour, and scavenging on risk of brucellosis transmission in elk of western Wyoming. J Wildl Dis 45:398–410

Mangas JG, Rodríguez-Estival J (2010) Logging and livestock influence the abundance of common mammal species in Mediterranean forested environments. Forest Ecol Manag 260:1274–1281

Margalida A, Donázar JA, Carrete M, Sánchez-Zapata JA, Donázar JA (2010) Sanitary versus environmental policies: fitting together two pieces of the puzzle of European vulture conservation. J Appl Ecol 47:931–935

Margalida A, Colomer MA, Sanuy D (2011a) Can wild ungulate carcasses provide enough biomass to maintain avian scavenger populations? An empirical assessment using a bio-inspired computational model. PLoS ONE 6:e20248

Margalida A, Campión D, Donázar JA (2011b) European vultures' altered behaviour. Nature 480:457

Margalida A, Carrete M, Sánchez-Zapata JA (2012) Good news for European vultures. Science 335:284

Markandya A, Taylor T, Murty MN (2008) Counting the cost of vulture declines: an appraisal of the human health and other benefits of vultures in India. Ecol Econ 67:194–204

McCarthy MA, Possingham HP (2007) Active adaptive management for conservation. Conserv Biol 21:956–963

Moreno-Opo R, Margalida A, Arredondo A, Guil F, Martín M, Higuero R, Soria C, Guzmán J (2010) Factors influencing the presence of the cinereous vulture *Aegypius monachus* at carcasses: food preferences and implications for the management of supplementary feeding sites. Wildl Biol 16:25–34

Naidoo V, Wolter K, Cuthbert R, Duncan N (2009) Veterinary diclofenac threatens Africa's endangered vulture species. Regul Toxicol Pharmacol 53:205–208

Oaks JL et al (2004) Diclofenac residues as the cause of vulture population decline in Pakistan. Nature 427:630–633

Olea P, Mateo-Tomás P (2009) The role of traditional farming practices in ecosystem conservation: the case of transhumance and vultures. Biol Conserv 142:1844–1853

Oro D, Margalida A, Carrete M, Heredia R, Donázar JA (2008) Testing the goodness of supplementary feeding to enhance population viability in an endangered vulture. PLoS ONE 3:e4084

Piper SE (2006) Supplementary feeding programs: how necessary are they for the maintenance of numerous and healthy vultures populations? In: Houston DC, Piper SE (eds) Proceedings of the international conference on conservation and management of vulture populations. Natural History Museum of Crete WWF, Greece, pp 41–50

Robb GN, McDonald RA, Chamberlain DE, Bearhop S (2008) Food for thought: supplementary feeding as a driver of ecological change in avian populations. Front Ecol Environ 6:476–484

Schiermeier Q (2001) Testing times for BSE. Nature 409:658–659

Şekercioğlu ÇH, Daily GC, Ehrlich PR (2004) Ecosystem consequences of bird declines. Proc Natl Acad Sci U S A 101:18042–18047

Shultz S et al (2004) Diclofenac poisoning is widespread in declining vulture populations across the Indian subcontinent. Proc R Soc Biol Sci 271:458–460

Sokal RR, Rohlf FJ (1995) Biometry: the principles and practice of statistics in biological research, 3rd edn. W. H. Freeman and Co., New York

StatSoft (2002) Statistica 6.1. Tulsa. www.statsoft.com. Accessed 11 March 2011

Tella JL (2001) Action is needed now, or BSE crisis could wipe out endangered birds of prey. Nature 410:408

Vicente J, Höfle U, Garrido JM, Fernández de Mera IG, Acevedo P, Juste P, Barral M, Gortázar C (2007) Risk factors associated with the prevalence of tuberculosis-like lesions in fenced wild boar and red deer in south central Spain. Vet Res 38:451–464

Virani MZ, Kendall C, Njoroge P, Thomsett S (2011) Major declines in the abundance of vultures and other scavenging raptors in and around the Masai Mara ecosystem, Kenya. Biol Conserv 144:746–752

Virgós E (2002) Factors affecting wild boar (*Sus scrofa*) occurrence in highly fragmented Mediterranean landscapes. Can J Zool 80:430–435

Whelan CJ, Wenny DG, Marquis RJ (2008) Ecosystems services provided by birds. Ann N Y Acad Sci 1134:25–60

Zuberogoitia I, Martínez JE, Margalida A, Gómez I, Azkona A, Martínez JA (2010) Reduced food availability induces behavioural changes in griffon vulture *Gyps fulvus*. Ornis Fenn 87:52–60

ESTADO DE CONSERVACIÓN

CONSERVATION STATUS

CAPÍTULO-CHAPTER
7

Conservation of the cinereous vulture in Spain (1966-2011): a bibliometric review of threats, research and adaptive management

by

Rubén Moreno-Opo & Antoni Margalida (*in revision*)

Conservación del buitre negro en España (1966-2011): revisión biobliométrica de amenazas, investigación y gestión adaptativa

La detección y cuantificación de amenazas, la investigación y la aplicación de medidas de gestión son aspectos clave para evaluar el estado de conservación de las especies amenazadas. Cuando dichas cuestiones pretenden ser analizadas con perspectiva histórica o a largo plazo, el análisis bibliométrico puede ser una técnica útil. Para el caso del buitre negro en España, se evaluó la relación existente entre la dinámica poblacional y los esfuerzos de investigación, las amenazas existentes y los hitos de conservación. Los resultados reflejaron una correlación de la tendencia poblacional positiva de la especie, de 206 parejas en 1976 a 2 068 en 2011, con el número total de trabajos, el número de publicaciones en revistas SCI y con los trabajos que trataron aspectos de conservación-amenazas-gestión. Dichas cuestiones se discuten en términos de causa-efecto teniendo en cuenta que otros factores no excluyentes entre sí probablemente también contribuyeron a explicar dicha tendencia poblacional. Del mismo modo, se analizaron las amenazas y la disponibilidad de alimento en relación al número de parejas reproductoras, obteniéndose una relación positiva con las estadísticas de caza mayor. En relación a los hitos, se concluye una situación actual positiva en cuanto a la protección de la especie y de su hábitat, no siendo clara la situación en relación a la disponibilidad de alimento. Por último, se repasan las principales líneas de actuación realizadas con la especie en España y cómo éstas han ido modulando progresivamente en función las nuevas evidencias científicas y técnicas, en un ejemplo de gestión adaptativa aplicada a conservación.

ABSTRACT
Detecting and quantifying threats, researching and implementing management actions are key factors to assess the conservation status of endangered species. When such questions are intended to be evaluated from long-term perspective, bibliometric analysis can constitute an useful tool. Taking as a case study the cinereous vulture *Aegypius monachus* in Spain, we tested relationships between population dynamics and research efforts, existing threats and conservation milestones. Our results suggest a positive correlation between the population increase of the species (from 206 pairs in 1976 to 2068 in 2011), with the total number of publications, the number of articles in SCI journals and the number of published works dealing with conservation-threats-management aspects. These results are discussed in terms of cause-effect relationships taking into account that influence of other non-mutually exclusive factors probably could also explain such associations. Similarly, we analyzed the trend of the cinereous vulture breeding population with respect to different threats and indexes of food available, obtaining a positive correlation with the big-game hunting bags progress in Spain. With respect to the conservation milestones, we concluded a positive current situation regarding the protection of the species and its habitat, not being clear the situation in relation to food availability. Finally, we reviewed the main conservation actions arranged with the species in Spain and how these progressively modulated on the basis of new scientific and technical evidences, as example of adaptive management applied to conservation.

KEY WORDS: animal by-products, bibliometry, conservation plans, habitat protection, land management, poisoning, Spain, special protected areas

INTRODUCTION
The assessment of the conservation status of endangered species requires several analysis stages, such as the recognition of the threats and their impacts, the monitoring of the population and the analysis of the efficiency of the applied management measures to eliminate risk factors (Soule 1986, Groom et al. 2006). These three aspects must be closely interrelated and without their proper coordination is very difficult to obtain successful results in conservation biology (Pullin et al., 2004, Arlettaz et al. 2010). When intending to study, with long time prospect, how these three questions have been conducted, in order to report on the history of conservation of a particular species, it is helpful to conduct an extensive collection of information, for which bibliometrics is a helpful study tool (i.e. Zhang et al. 2010, Liu et al. 2011). In this sense, assuming that the increase in the knowledge of a target species and the application of such information to optimize management and conservation measures will imply to improve their conservation status, we hypothesized that their population trends should be parallel to the increase of published literature. As a result, the analysis of publications may establish relationships of the research effort and the field study with variables such as the population trend of the species, the non-natural mortality rates or the conservation measures implemented, allowing the discussion about the relations between them. Obviously, it will exist cases in which both, the positive or negative effect of anthropogenic variables (i.e. illegal

poisoning, disturbance, habitat alterations, supplementary feeding) can have an important influence on the species population trend and as consequence distort the linearity of the results obtained (Oro et al. 2008, Ortega et al. 2009). However, the expected result is that conservation measures and for extension the conservation status of a species will improve with an increase in their knowledge.

To analyze the above issues, the cinereous vulture *Aegypius monachus* constitutes a sound model species since it exhibits a key role in balancing ecosystems it inhabits. Due to its scavenger behaviour, it provides important ecosystem services: it feeds on wildlife and livestock carcasses and helps to reduce the risks of spread of transmissible diseases (DeVault et al. 2003, Sekercioglu et al 2004, Margalida et al. 2012, Ogada et al. 2012). It also inhabits areas in good conservation status and is indicator of several characteristics of the habitat: to breed it requires mature forests, with specified ecological and geomorphologic conditions, far away from disturbing human activities (Moreno-Opo et al. 2012a). Spatial patterns of its foraging ranges are related to food availability and habitat quality and, in the case of breeding individuals, to the distance from the breeding colony (Costillo 2004, Carrete and Donázar 2005). It is also closely related to the land management and exploitation made in rural areas which can affect in positive or negative ways (Donázar et al. 2002, 2009a).

This paper presents the conservation story of the cinereous vulture in Spain during 1966-2011 in a summarized way, as a compilation of studies and conservation works. Its objectives are: 1) to update the census of the species in Spain and to evaluate its population trend, 2) to gather information on the researching, monitoring and protection efforts made in Spain, 3) to assess the relationships between the number and types of publications and the population dynamics of the species in order to discuss if a greater study output is cause or consequence of the numerical evolution of the population, 4) to analyze the effects of threats to the cinereous vulture, as well as the actions taken and their results, from a perspective of conservation biology.

METHODS

The study species

The cinereous vulture is the largest raptor of the Palearctic and its distribution range includes temperate latitudes from the Iberian to the Korean peninsulas (Del Hoyo et al. 1994). Its current population has been estimated in 7 200-10 000 pairs (BirdLife International 2008), of which around 2 000 are located in Europe (BirdLife International 2004). It is considered Near Threatened due to the decline of its Asian populations and to different threatening factors: mortality caused by human actions (mainly by the use of illegal poisoned baits), alteration of food availability and occurrence, human disturbances and habitat loss (BirdLife International 2008). In Europe, the cinereous vulture is distributed within three metapopulations: Iberian Peninsula and France, Majorca island and Balkan Peninsula (Del Hoyo et al. 1994). The Spanish population is the greater (De la Puente et al. 2007) and more studied in recent years (i.e. Costillo 2004, Carrete and Donázar 2005, Morán-López et al. 2006, Moreno-Opo et al. 2012a), exhibiting during the last 40 years a continuous recovery (De la Puente et al. 2007). It is one of the most emblematic species of the Mediterranean forest, having received a soundly number of monitoring, researching and protection actions by governments and scientific and conservation organizations (Moreno-Opo and Guil 2007, De la Puente 2012, Dobado and Arenas 2012).

Bibliographic search and studied variables

The breeding pairs, as the number of nests in which incubation started, was obtained through the request to the regional governments of the most updated census. Precedent census were consulted in compiling publications (i.e. González 1990, Sánchez 1998, De la Puente et al. 2007).

With the aim of finding published works in which the cinereous vulture in Spain was targeted we performed a literature search (see Liu et al. 2011). We used searchers (Web of Science, Google Scholar), reviewed references in the articles, and made a compilation of books, PhD dissertations, legal texts and abstracts in proceedings of meetings on the cinereous vulture. The works were grouped according to: 1) the type and/or character of the publication in which the work appeared, and 2) the topic of the work. For the section 1, works were assigned to the following five categories: SCI journals, other technical-scientific articles in periodic journals, legal-divulgation, books-PhD thesis, and chapters-articles in abstracts of monographs, workshops or congresses. For the part 2, works were grouped as the field study dealt: census, biology-ecology, conservation-threats-management, and others that included shared aspects to the above categories, compilations, distribution atlas, etc. Furthermore, works were clustered, according to their year of publication, into five-year periods from 1966 to 2011 to analyze their trend.

To assess the effect of different threats on the population dynamics of the cinereous vulture, we considered the number of individuals affected by poisoning, obtained from Hernández and Margalida (2008), and the number of admissions into official rescue centres due to starvation or dehydration. This latter information was provided by 10 regional administrations for the period 2001-2009 (Ministry of Environment and Rural and Marine Affairs 2010). Moreover, we used variables related to the availability of food: 1) the hunting bags of the wild rabbit *Oryctolagus cuniculus* in the regions of peninsular Spain where cinereous vulture breeds (Guil et al. 2007, Garrido 2011), as an index of relative abundance of one of the main prey in the diet of the species (Costillo et al. 2007a), 2) the hunting statistics of the two most abundant game ungulates -wild boar *Sus scrofa* and red deer *Cervus elaphus*- in the regions of peninsular Spain where cinereous vulture breeds (Garrido 2011), as the remains of the hunting activity on these species are traditionally intended to scavengers consumption and are an important source of the diet of the cinereous vulture (Costillo et al. 2007a, Moreno-Opo and Guil 2007) and, 3) the number of extensive livestock of bovine, ovine, caprine and porcine not integrated in insurances programs of removal of carcasses, as an index of the trophic resources potentially available in the wild for scavenger raptors -by deducting the proportion of livestock covered by official corpses-collection insurances from the total extensive livestock population- (National Entity of Agriculture Insurances 2011, Spanish Statisticts Institute 2011).

Lastly, we performed a review of the most important events for the conservation of the cinereous vulture in Spain during the study period (1966-2011). These milestones were related to the release of legal and technical crucial documents or to biological situations relevant to three important issues for the conservation of the species (BirdLife International 2008): species protection, habitat protection and food availability.

We chose 1966 as the start of this study since it is the first year with scientific and technical publications on the cinereous vulture in Spain (Bernis 1966, Suetens and Van Groenendal 1966, Valverde 1966).

Statistical analyses

We first checked differences in the distribution of the published papers into different five-periods, regarding the topic discussed and the type of publication. Thus, we performed Chi-square frequency analysis.

In order to correlate the population trend in Spain during 1966-2011, we performed regression analysis (Sokal and Rohlf 1995) between the number of breeding pairs and 1) bibliometric variables – total number of publications and number of published works classified by type of publication and by subject treated-, 2) variables related to direct threats – number of cinereous vultures found poisoned and number of starved individuals admitted to rescue centres- and 3) variables related to the potential food availability –wild rabbit hunting statistics, potential number of available carcasses from extensive livestock, big-game hunting bags, and the previous three variables together-. On the other hand, to assess possible correlations between food availability and threats, we applied simple regression analysis between the number of potential available livestock corpses and 1) the number of poisoned cinereous vultures and, 2) the number of starved/dehydrated cinereous vultures admitted to rehabilitation centres. The analyses were carried out using the software Statistica 6.1 (StatSoft Inc. 2002).

RESULTS

The breeding population of cinereous vulture in Spain in 2011 amounted 2 068 pairs, distributed in 35 colonies (Fig. 1). Since 1973, when the first national census yielded 206 breeding pairs, there has been an annual increase rate of 25.7% in the number of breeding pairs (Fig. 1).

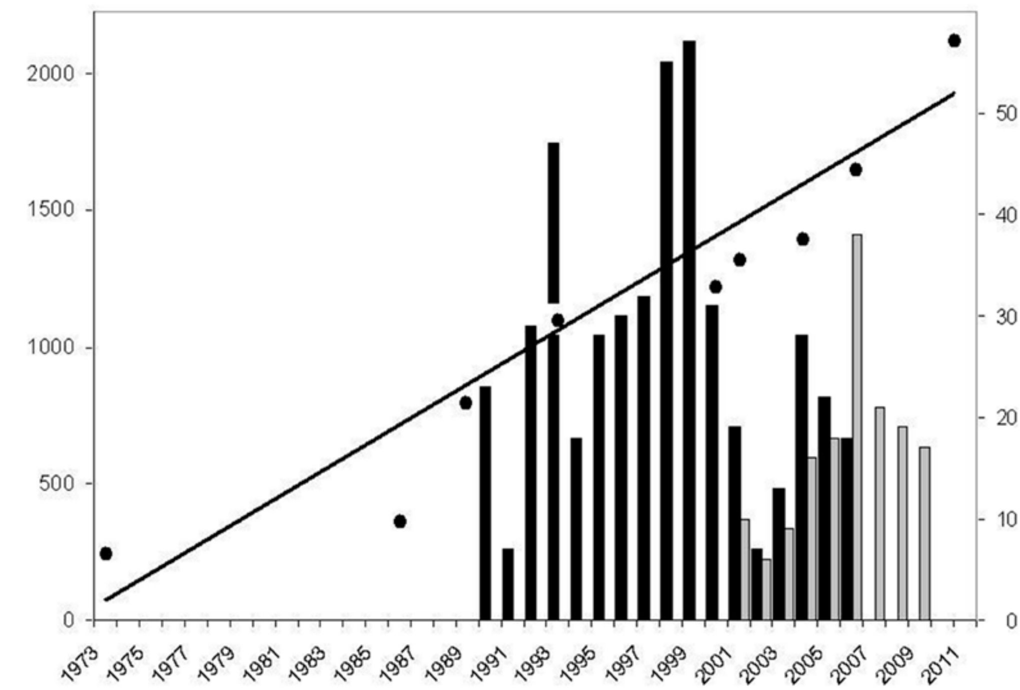

Figure 1. Breeding pairs (nests with hatching, black dots, left y-axis) of cinereous vulture *Aegypius monachus* in Spain, since the first Spanish national census in 1973 to the most updated in 2011. The number of poisoned cinereous vulture (1990-2006, black columns, Hernández and Margalida 2008) and the number of starved cinereous vultures admitted to official wildlife rescue centres in (2001-2009, grey columns) are shown respect the right y-axis.

In the period 1966-2011, 164 papers on the cinereous vulture in Spain were published (Appendix S1). Regarding the type of publication, 31.5% were articles in technical and scientific periodical journals not included in SCI, 30.9% were legal or divulgative releases, 17.5% papers in SCI journals, 14.0% abstracts in proceedings of conferences or chapters of monographs, and 5.8% books or PhD thesis. The proportion of the different literature published varied significantly among the different studied five-year periods (χ^2 = 11.48; df = 4; p = 0.021; Fig. 2). In relation to the topics of the paper, 32.3% dealt with conservation, threats or applied management, 31.8% presented aspects of the biology and ecology of the species, 21.9% population census and 13.8% exposed common issues to the above mentioned categories or were compilations, atlas, etc. No significant differences were found with respect the proportion of the subjects treated between different periods (χ^2 = 4.11; df = 3; p = 0.249; Fig. 2).

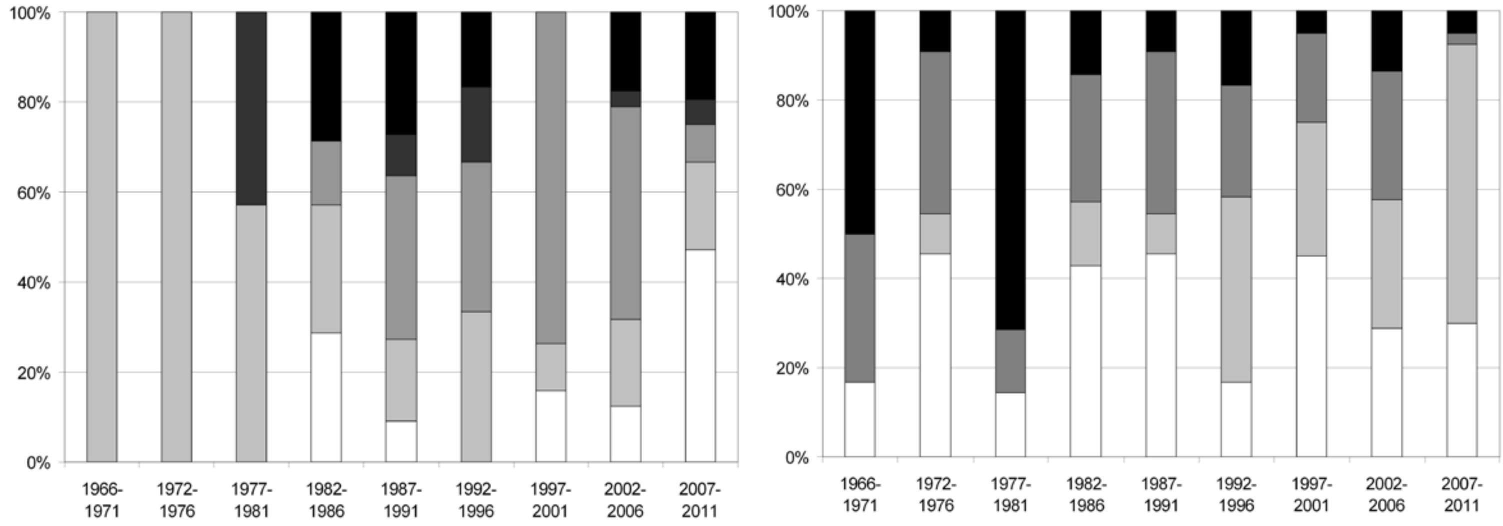

Figure 2. Evolution of the proportion of publications (Y-axis) on the cinereous vulture *Aegypius monachus* in Spain in five-year periods, distributed according to the type of publication (left; SCI journals –white-, other technical-scientific articles in periodic journals –light grey-, legal-divulgation –medium grey-, books-PhD thesis –dark grey- and chapters-articles in abstracts of monographs, workshops, congresses or meetings –black-) and the subject treated (right; biology-ecology –white-, conservation-threats-management –light grey-, census –dark grey- and others –black-)

The unique common Spanish database of wildlife poisonings showed the affection to 454 cinereous vultures during the period 1990-2006, with a maximum in 1998 and 1999 (Hernández and Margalida 2008; Fig. 1). Data of admissions of cinereous vultures in rescue centres showed and increasing trend from 2001 to 2007, when a maximum of 190 entrances were recorded, which was subsequently reduced to 2009 (Fig. 1). The analysis of the most relevant historical events (Fig. 3) provided that current situation is positive with respect to the protection of the species and its habitat, while there is no certainty about whether the food occurrence and availability is favourable or unfavourable during the last decade.

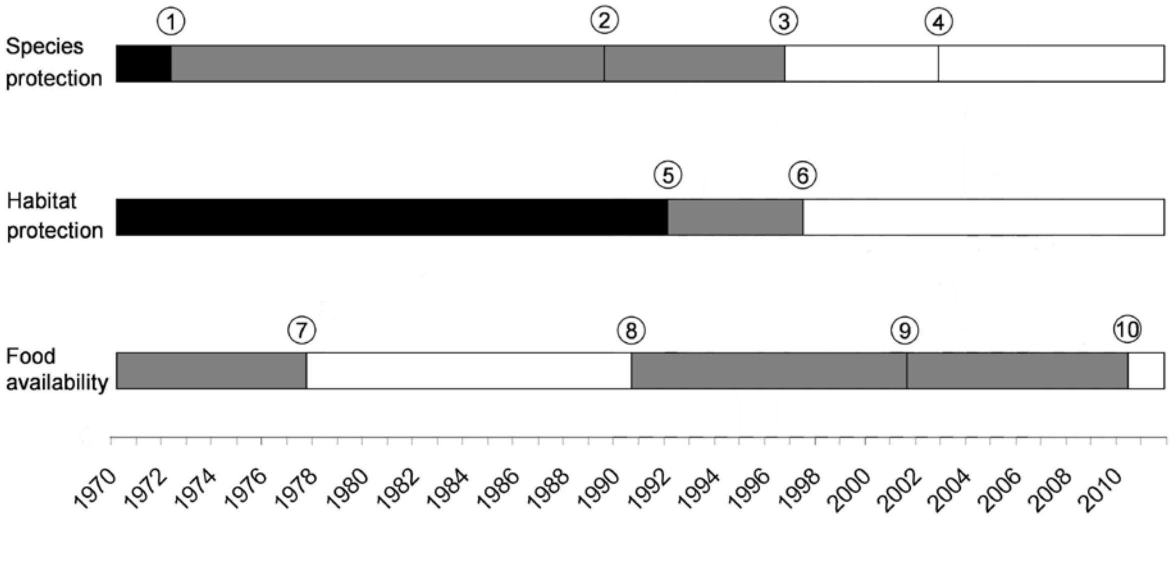

Figure 3. Milestones of the conservation of the cinereous vulture *Aegypius monachus* in Spain in 1970-2011, in relation to the species protection, the habitat protection and the food availability. Colours of horizontal bars indicate the conservation status in relation to the ecological requirements of the species (black: negative situation; grey: unknown-neutral; white: favourable situation). 1= Decree for the protection of raptors in Spain -1973-; 2= Royal Decree of the National Catalogue of Endangered Species -1990-; 3= Species Action Plan, European Commission -1996-; 4= First regional recovery/conservation plans -2003-; 5 = Habitats Directive 92/43/CEE -1992-; 6= Special Protected Areas statement and first LIFE and land-stewardship projects -1997-; 7= Progressive recovery of wild rabbit populations after myxomatosis outbreak in the 50´s –1978-; 8= Outbreak of the rabbit haemorrhagic disease – 1991-; 9= Regulation CE 1774/2002 on the sanitary control of animal by-products -2002-, but progressive increase of wild boar *Sus scrofa*/red deer *Cervus elaphus* hunting bags; 10= Regulation CE 1069/2009 on the sanitary control of animal by-products -2011-.

Since 2002 there was a progressive and significant reduction in the number of available carcasses in the wild, following the entry into force of the mandatory carcasses collection (Council of Europe 2002), from 36 million potential available carcasses in 2001 to 3.6 in 2010 ($r = -0.682$; $p < 0.001$; Fig. 4). Furthermore, the number of hunted wild boar and red deer per year has been multiplied by 13.5 from 1973 ($r = 0.94$; $p < 0.001$; Fig. 5). Furthermore, wild rabbit hunting statistics have showed variations without any clear trend ($r = -0.11$; $p = 0.789$). After a slight abundance increase until the late 80s a shocking decrease occurred that has been attenuated and even reverted since 2008 (Fig. 4).

The trend of breeding pairs of cinereous vulture was positively correlated with the total number of published studies ($r = 0.757$; $p = 0.048$). Likewise, the number of breeding pairs was positively related to the number of publications in SCI journals ($r = 0.824$; $p = 0.022$), to the number of papers that addressed conservation and management of threats ($r = 0.909$; $p = 0.004$), and, marginally, to works on ecology and biology ($r = 0.749$; $p = 0.052$). There were no statistically significant relationships ($p > 0.05$) of the trend of the breeding population with other classes of publications or to the number of poisoned/entered to rescue centres cinereous vultures. There was also no correlation between cinereous vulture population with food availability variables, except for the number of wild boar and red deer hunted ($r = 0.984$; $p < 0.001$). Finally, the admissions of starved cinereous vultures into recovery centres were negatively and marginally correlated with the decrease of available carcasses ($r = -0.926$; $p = 0.068$).

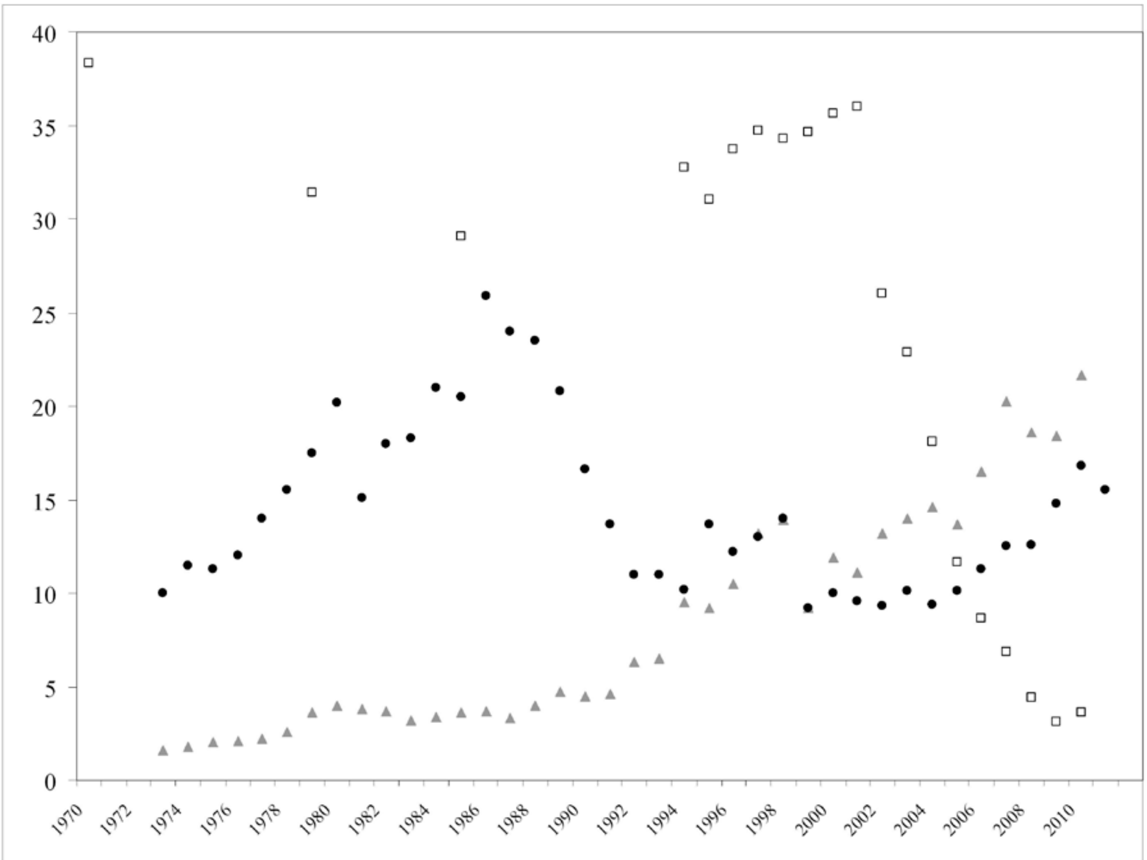

Figure 4. Evolution of hunted wild rabbits *Oryctolagus cuniculus* in the Spanish peninsular regions where cinereous vulture *Aegypius monachus* breeds (black dots), the number of livestock carcasses potentially available for scavengers in Spain (x10^6; squares), and wild boar *Sus scrofa* and red deer *Cervus elaphus* hunting bags in the Spanish peninsular regions where cinereous vulture *Aegypius monachus* breeds (x10^4; grey triangles). For the wild rabbit, the number of hunted individuals in 1973 is taken as reference (starting value = 10.0, not corresponding to any unit, Guil et al. 2007, Garrido 2011). The number of carcasses available has been obtained by deducting the proportion of livestock covered by official corpses-collection insurances from the total extensive livestock population (sources; Spanish Statisticts Institute www.ine.es and National Entity of Agriculture Insurances, 2011).

DISCUSSION

This work constitutes the first long-term study about the association between published research and conservation of the cinereous vulture. However, this information includes different deficits which require a cautious interpretation regarding conclusions about cause-effect relationships. This is the case of the use of a single variable (number of breeding pairs) to assess the population trend as consequence of the difficult to obtain extensive data of other demographic parameters as the non-breeding proportion of the population, the total number of pairs, mortality rates, breeding success, etc. that could help to understand and to analyze more accurately the population trend (Oro et al. 2008, Ortega et al. 2009). In this sense, a common problem with census of elusive species is that their reliability improves with time due to a progressively better knowledge of the terrain and the species, as well as due to an increasing investment in logistics. This could imply variable census effort between years and areas, and negative effects in the accuracy and bias of the final data of population size (Katzner et al. 2011, Margalida et al. 2011a).For this reason we used the comparison between censuses due to their historic basis, for being the only ones available and after knowing that all or most of the existing colonies were prospected in each census.

Threats

In relation to the food availability, a direct relationship between the increase of the breeding population of the cinereous vulture and big-game hunting bags was evidenced, since this leisure activity has experienced an important rise in Spain (Garrido 2011). On the other hand, changes in animal by-products regulations occurred in early of the 2000s, that led to the compulsory collection of livestock carcasses from the wild provoking food shortages (Donázar et al. 2009a, 2009b), have not been associated with a reduction of the breeding population of the cinereous vulture. On the other hand, wild rabbit, a keystone prey-species for most endangered predators in Spain (Delibes-Mateos et al. 2007), has suffered an important population decline during the last 40 years, provoking dietary shifts in Spanish cinereous vulture population (Costillo et al. 2007a, b, Corbacho et al. 2007). This dietary plasticity and the adaptation to feed on alternative preys when the main food resource is scarce have been evidenced in the vulnerable Spanish imperial eagle after the decrease of wild rabbit abundance (Sánchez et al. 2009) and in other obligate avian scavenger as consequence of changes in sanitary policies (Donázar et al. 2010). Nevertheless, other effects have been reported in the scavenger guild regarding demographic and behavioural variables (e.g. modification of the foraging, and dispersal patterns, trophic behaviour increasing feeding on dumps and attacks on livestock, changes in juvenile survival rates, Donázar et al. 2009b, Margalida et al. 2010, Zuberogoitia et al. 2010, Margalida et al. 2011b).

Legal protection of breeding areas, reaching more than 90% of the breeding pairs in Spain (Moreno-Opo and Guil 2007), ensured the protection of wide territories by preventing actions irreversibly altering natural habitats. From the 80s and, above all, 90s a large number of protected natural areas and special protected areas for birds (Council of Europe 1992) were declared and several official conservation plans were adopted. This fact has succeeded in minimizing the negative effects of human disturbance during the breeding season and in reducing habitat loss, despite some difficulties in managing these areas in relation to reconciling economic uses in them (Margalida et al. 2011c).

It is possible that other factors may have influenced the cinereous vulture population trend. Issues such as socio-cultural changes in the perception of nature conservation occurring in Spain, the inclusion into governmental bodies of technicians and biologists trained in wildlife protection, changes in rates of economic progress, development of infrastructure, etc., may also correlate with the population dynamics of the species.

Research and population trend

The bibliometric analysis showed how papers published in SCI journals, as well as studies on conservation, threats and management issues and, to a lesser extent, works dealing with the species biology grew up with a similar trend as the breeding population of the cinereous vulture in Spain. This could highlight that a better understanding of the ecological requirements and the magnitude of the threats of the studied species may contribute to a greater success in conservation policies (Pullin et al. 2004, Sutherland et al. 2004, Arlettaz et al. 2010) and, consequently, an improvement in the situation of the species population. However, it is difficult to assess whether there exists causal link between the study effort and conservation status or population trend of the targeted species (Pullin and Stewart 2006). To this constraint it could contribute several facts, as the possibility that it is unknown *a priori* whether the studied matter is really useful and important to address the risk factors of the species, if there are other major non-weighted or difficult-to-investigate threats, whether results obtained in a geographical and temporal framework are representative and applicable to other regions or circumstances, or if the management recommendations are feasible, rather expensive or require highly qualified training to be implemented (Pullin and Knight 2005, Arlettaz et al.2010).

In addition, in order to justify causality, it might be necessary a time mismatch between the publication of the work, its application and the verification of positive effects. Thus, it could be argued that publication of valuable scientific works comes after the release of an adequate basic knowledge of the species (Faaborg et al. 2010) and, even, as a result of the recovery of its populations. This would allow to deepen into up-to-date, fashionable study lines or to increase the efficiency of results, as a consequence of a likely accessibility to the species, a greater sample size or a greater ease for obtaining official permits. Therefore, core, accurate and well-publicized information on the species may be as important for the conservation of endangered species such as the publication of scientific studies in SCI journals (Martín-López et al. 2009, Botrill et al. 2011). In this sense, several examples on well studied targeted species exist in which the increase in their knowledge has not been parallel to their recovery. For example, the California condor *Gymnogyps californianus* in the USA that reduced their numbers reaching the near extinction as consequence of lead poisoning mortality (Finkelstein et al. 2012) or the Egyptian vulture *Neoprhon percnopterus* that in the last decades reduced their population in some Spanish areas as consequence of the illegal use of poisoning (Carrete et al. 2007, Hernández and Margalida 2009). In both cases, the increase of non-natural anthropogenic mortality factors nor detected or minimized their impact previously provoked failures in the application of conservation measures.

Anyway, the exercise of *stop and assess* the results of scientific research, the legal and economic endowments for the implementation of practical actions and the population dynamics of an endangered species, with somewhat historical perspective, is very relevant to conclude whether the decisions adopted have been successful or should be modified (Salafsky et al. 2001, McCarthy and Possingham 2007, Palomares et al. 2011).

Adaptive management

The cinereous vulture in Spain could provide an example of *coevolution* between the recognition of threats and the application of management actions on them. First, in the 70 and 80 decades the need for the legal protection of the species was advertised. So, this issue was subsequently implemented with the enactment of different acts and conservation plans (i.e. Ministry of Agriculture 1990, Heredia 1996, Fig. 5). Later, in the 80s-90s, the lack of proper habitat protection conditioned the safety of the species mainly in breeding territories. This situation led to the statement of the Special Protected Areas for birds (Council of Europe 1992) and to the implementation of demonstrative projects on best land management practices.

During the first decade of the twentieth century, progresses have been made on the detailed study of limiting factors such as the food availability and occurrence, the mortality due to illegal poisoning, the compatible management of the exploitation of natural resources or the settlement of new breeding areas (Moreno-Opo and Guil 2007, Donázar et al. 2009a, Del Moral and De la Puente 2010). In this regard, best available knowledge has been engaged to adapt the early management policies into the most appropriate needs of the species (McCarthy and Possingham 2007). As a result, for instance, 1) a new legal framework is being enforced to enable cinereous vulture to feed in an extensive and sustainable way, compared to the previous sole possibility through fenced supplementary feeding points (European Commission 2011, Margalida et al. 2012, Moreno-Opo et al. 2012b); 2) the procedures of forestry activities in breeding areas of cinereous vulture have been evaluated promoting, when appropriate, the substitution of the ban of certain activities for its implementation according to defined technical protocols (Junta de Extremadura 2005, Moreno-Opo and Guil 2007, Margalida et al. 2011c); 3) the knowledge of harmful chemical products used in phytosanitary treatments has increased, and their prohibition and replacement by other compounds has been fostered (Hernández and Margalida 2008, Council of Europe 2009); 4) monitoring methods of the cinereous vulture have been analyzed, suggesting a homogenization of data collection that allows comparison of results among different Spanish regions (De la Puente et al. 2007); 5) it has been highlighted the need for protection and suitable management of the foraging habitat, as the geographic scope in which many of the threats of the species are recorded, in relation to the initial approach that just prioritized the nesting habitat (Carrete and Donázar 2005, Moreno-Opo et al. 2010); 6) the principles of operating against the illegal use of poisoned baits have been updated, incorporating informative and practical preventive measures to surveillance and punishment (Conover 2001, Dirección General para la Biodiversidad 2007); and 7) it has been indicated the usefulness of common databases at the national or international level on different incident threats, especially poisoning, as evidence base for prioritizing conservation measures (Mateo 2010).

Although its population trend is expected to be favourable, it is necessary to continue monitoring, studying and to maintaining conservation efforts on the cinereous vulture in Spain. This will, first, ensure the ecosystem services provided by the species, especially those from the recycling of carcasses in an economical and hygienic way (DeVault et al. 2003; Margalida et al. 2012) and those from its role as indicator of the conservation status of the environments (Moreno-

Opo et al. 2012a). Similarly, and since Spain hosts about 20-25% of the whole breeding pairs (BirdLife International 2008), this population may become a guarantee of the global conservation of the species versus the decline events in Asian populations (BirdLife International 2008) and source to promote the reintroduction and connection of the different metapopulations (Houston 2006).

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REFERENCES

Arlettaz, R., Schaub, M., Fournier, J., Reichlin, T. S., Sierro, A., Watson, J. E. M. and Braunisch, V. (2010). From Publications to Public Actions: When Conservation Biologists Bridge the Gap between Research and Implementation. *Bioscience* 60: 835–842.

Bernis, F. (1966). El buitre negro *Aegypius monachus* en Iberia. *Ardeola* 12: 45-99.

BirdLife International. (2004). *Birds in Europe: population estimates, trends and conservation status.* BirdLife International. Cambridge.

BirdLife International. (2008) *Aegypius monachus*. In: *UICN Red List of Threatened Species*. Version 2011.2. www.iucnredlist.org. Downloaded on 27 March 2012.

Botrill, M. C., Walsh, J. C., Watson, J. E. M., Joseph, L. N. Ortega-Argueta, A. and Possingham, H. P. (2011). Does recovery planning improve the status of threatened species? *Biological Conservation* 144: 1595-1601.

Carrete, M. and Donázar, J. A. (2005). Application of central-place foraging theory shows the importance of Mediterranean dehesas for the conservation of cinereous vulture *Aegypius monachus*. *Biological Conservation* 126: 582-590.

Carrete, M., Grande, J.M., Tella, J.L., Sánchez-Zapata, J.A., Donazar, J.A., Díaz-Delgado R. and Romo, A. (2007). Habitat, human pressure, and social behavior: partialling out factors affecting large-scale territory extinction in an endangered vulture. *Biological Conservation* 136: 143-154.

Conover, M. R. (2001). *Resolving human-wildlife conflicts. The science of wildlife damage management*. Lewis Publishers. New York.

Corbacho, C., Costillo, E. and Perales, A. B. (2007). Feeding of the cinereous vulture. In: Moreno-Opo, R. and Guil, F. (eds.) *Handbook on the management of the habitat and populations of the cinereous vulture in Spain.* Ministry of Environment, pp. 179-196. Madrid. (in Spanish).

Costillo, E. (2004). *Biología y conservación de las poblaciones de buitre negro Aegypius monachus en Extremadura*. PhD Thesis. University of Extremadura.

Costillo, E., Corbacho, C., Morán, R. and Villegas, A. (2007a). The diet of black vulture *Aegypius monachus* in response to environmental changes in Extremadura (1970 - 2000). *Ardeola* 54: 197-204.

Costillo, E., Corbacho, C., Morán, R. and Villegas, A. (2007b). Diet Plasticity of Cinereous Vulture *Aegypius monachus* in Different Colonies in the Extremadura (SW Spain). *Ardea* 95: 201-211.

Council of Europe. (1992). Directive 43/92/CEE, of 21 May 1992, on the conservation of natural habitats and of wild fauna and flora. *Official Journal of the European Union* 206: 1-66.

Council of Europe. (2002). Regulation CE 1774/2002 of the European Parliament and the Council of 3rd October. *Official Journal of the European Union* 273: 1-154.

Council of Europe. (2009). Regulation CE 1107/2009 of the European Parliament and the Council of 21st October, concerning the placing of plant protection products on the market and repealing Council Directives 79/117/EEC and 91/414/EEC. *Official Journal of the European Union* 309: 1-50.

De la Puente, J., Moreno-Opo, R. and Del Moral, J. C. (2007) *El buitre negro en España: censo nacional 2006*. SEO/BirdLife. Madrid.

De la Puente, J. (in press). *El buitre negro en el Parque Natural de Peñalara*. Comunidad de Madrid. Madrid.

Del Hoyo, J., Elliott, A. and Sargatal, J. (1994). *Handbook of the Birds of the World*. 2. Lynx Edicions. Barcelona.

Del Moral, J. C. and De la Puente, J. (2010). Buitre negro *Aegypius monachus.* In: Salvador, A. and Carrascal, L. M. (Eds). *Enciclopedia virtual de los vertebrados españoles*. Museo Nacional de Ciencias Naturales. Madrid. http://www.vertebradosibericos.org/aves/aegmon.html

Delibes-Mateos, M., Redpath, S.M., Angulo, E., Ferreras, P. and Villafuerte, R. (2007) Rabbits as a keystone species in southern Europe. *Biological Conservation* 137: 149-156.

DeVault, T. L., Rhodes, O. E. and Shivik, J. A. (2003) Scavenging by vertebrates: behavioural, ecological, and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. *Oikos* 102: 225-234.

Dirección General para la Biodiversidad. (2005). *National Strategy against the illegal use of poisoned baits in the wild*. Ministry of Environment. Madrid.

Dobado, P. and Arenas, R. (2012) *TheBlack Vulture: Status, Conservation and Studies.* Consejería de Medio Ambiente de la Junta de Andalucía. Córdoba.

Donázar, J. A., Blanco, G., Hiraldo, F., Soto-Largo, E. and Oria, J. (2002). Effects of forestry and other land-use practices on the conservation of cinereous vultures. *Ecological Applications* 12: 1445–1456.

Donázar, J.A., Margalida, A.and Campión, D. (2009a).Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology. *Munibe* 29. Sociedad de Ciencias Aranzadi. Donostia.

Donázar, J. A., Margalida, A., Carrete, M., Sánchez-Zapata, J. A. (2009b) Too sanitary for vultures. *Science* 326: 664.

European Commission. (2011). Commission Regulation CE 142/2011, of 25th February, implementing Regulation (EC) No 1069/2009 (…) *Official Journal of the European Union* 54: 1-254.

Faaborg, J. et al. (2010). Conserving migratory land birds in the New World: Do we know enough? *Ecological Applications* 20: 398-418.

Finkelstein, M.E., Doak, D.F., George, D., Burnett, J., Brandt, J., Church, M., Grantham, J. and Smith, D.R. (2012). Lead poisoning and the deceptive recovery of the critically endangered California condor. *Proceedings of the National Academy of Sciences of the USA* doi: 10.1073/pnas.1203141109

Garrido, J. L. (2011). *Estimación de aprovechamientos cinegéticos en España por especies y comunidades autónomas.* Fedenca. Madrid.

González, L. M. (1990). Situación de las poblaciones de águila imperial y buitre negro. *Quercus* 58: 16-22.

Guil, F., Agudín, S., Silvestre, F., Higuero, R. and Inogés, J. (2007). Small game and cinereous vulture. Promoting wild rabbit populations. In: Moreno-Opo, R. and Guil, F. (eds.) *Handbook on the management of the habitat and populations of the cinereous vulture in Spain.* Ministry of Environment, pp. 224-253. Madrid. (in Spanish).

Groom, M. J., Meffe, G. K. and Carroll, C. R. (2006). *Principles in Conservation Biology. Third Edition*. Sinauer Associates, Inc. Sunderland.

Heredia, B. (1996). Action Plan for the Cinereous Vulture (*Aegypius monachus* in Europe. In: Heredia, B., Rose, L. and Painter, M. (eds.) *Globally threatened birds in Europe: action plans*. Council of Europe-BirdLife International, pp. 147-158. Strasbourg.

Hernández, M. and Margalida, A. (2008). Pesticide abuse in Europe: effects on the cinereous vulture population in Spain. *Ecotoxicology* 17: 264-272.

Hernández, M. and Margalida, A. (2009). Poison-related mortality effects in the endangered Egyptian vulture (*Neophron percnopterus*) population in Spain. *European Journal of Wildlife Research* 55: 415-423.

Hiraldo, F. (1983). Breeding biology of the Cinereous Vulture. In: Wilbur, S. R. and Jackson, J. A. (Eds). *Vulture Biology and Management*, pp. 197-213. University of California Press. Berkeley.

Houston, D. C. (2006). Reintroduction Programmes for Vulture Species. Houston, D.C. and Piper, S. E. (eds). *Proceedings of the International Conference on Conservation and Management of Vulture Populations*, pp. 87-97. Natural History Museum of Crete-WWF Greece. Thessaloniki.

Junta de Extremadura. (2005). Orden de 6 de junio de 2005 por la que se aprueba el Plan de Conservación del Hábitat del Buitre Negro en Extremadura. *Diario Oficial de Extremadura* 71: 8877-8886.

Katzner, T.E., Ivy, J.A.R., Bragin, E.A., Milner-Gulland, E.J. and DeWoody, J.A. (2011) Conservation implications of inaccurate estimation of cryptic population size *Animal Conservation* 14: 328–332

Liu, X., Zhang, L. and Hong, S. (2011). Global biodiversity research during 1900–2009: a bibliometric análisis. *Biodiversity and Conservation* 20:807–826.

Margalida, A., Heredia, R., Razin, M. & Hernández, M. (2008). Sources of variation in mortality of the Bearded Vulture *Gypaetus barbatus* in Europe. *Bird Conservation International* 18: 1-10.

Margalida, A., Donázar, J. A., Carrete, M. and Sánchez-Zapata, J. A. (2010). Sanitary versus environmental policies: fitting together two pieces of the puzzle of European vulture conservation. *Journal of Applied Ecology* 47: 931-935.

Margalida, A., Oro, D., Cortés-Avizanda, A. Heredia, R. & Donázar, J.A. (2011a). Misleading population estimates: biases and consistency of visual surveys and matrix modelling in the endangered bearded vulture. *PLoS ONE* 6: e26784

Margalida, A., Campión, D. and Donázar, J. A. (2011b). European vultures´ altered behaviour. *Nature* 480: 457.

Margalida, A., Moreno-Opo, R., Arroyo, B. E. and Arredondo, A. (2011c). Reconciling the conservation of an endangered species with an economically important anthropogenic activity: interactions between cork exploitation and the Cinereous Vulture *Aegypius monachus* in Spain. *Animal Conservation* 14: 167–174.

Margalida, A., Carrete, M., Sánchez-Zapata, J. A. and Donázar, J. A. (2012). Good News for European Vultures. *Science* 335: 284.

Martín-López, B., Montes, C., Ramírez, L. and Benayas, J. (2009). What drives policy decision-making related to species conservation? *Biological Conservation* 142: 1370-1380.

Mateo, R. (2010). Toxicology and wildlife conservation in Europe: the inadequacy of current EU regulations. *The Veterinary Journal* 183: 241-242.

McCarthy, M. A. and Possingham, H. P. (2007) Active Adaptive Management for Conservation. *Conservation Biology* 21: 956–963.

Ministry of Agriculture. (1990). Royal Decree 439/1990,regulating the National Catalogue of Endangered Species. *Boletín Oficial del Estado* 82: 9468-9471.

Ministry of Environment, and Rural and Marine Affairs. (2010). *Situation of the populations of scavenger raptors in relation to food availability*. Unedited report. Madrid.

Morán-López, R., Sánchez, J. M., Costillo, E., Corbacho, C. and Villegas, A. (2006) Spatial variation in anthropic and natural factors regulating the breeding success of the cinereous vulture *Aegypius monachus* in SW Iberian Peninsula. *Biological Conservation* 130: 169–182.

Moreno-Opo, R. and Guil, F. (2007) *Handbook on the management of the habitat and populations of the cinereous vulture in Spain.* Ministry of Environment. Madrid. (in Spanish).

Moreno-Opo, R., Arredondo, A. and Guil, F. (2010). Foraging range and diet of cinereous vulture *Aegypius monachus* using livestock resources in central Spain. *Ardeola* 57: 111-119.

Moreno-Opo, R., Fernández-Olalla, M., Margalida, A., Arredondo, A. and Guil, F. (2012a). Effect of Methodological and Ecological Approaches on Heterogeneity of Nest-Site Selection of a Long-Lived Vulture. *PLoSONE* 7(3): e33469.

Moreno-Opo, R., Margalida, A., García, F., Arredondo, A., Rodríguez, C. and González, L. M. (2012b). Linking sanitary and ecological requirements in the management of avian scavengers: effectiveness of fencing against mammals in supplementary feeding sites. *Biodiversity and Conservation* 21: 1673-1685.

Ogada, D. L., Torchin, M. E., Kinnaird, M. F. and Ezenwa, V. O. (2012). Effects of Vulture Declines on Facultative Scavengers and Potential Implications for Mammalian Disease Transmission. *Conservation Biology* DOI: 10.1111/j.1523-1739.2012.01827.x

Oro, D., Margalida, A., Carrete, M., Heredia, R. and Donázar, J.A. (2008). Testing the Goodness of Supplementary Feeding to Enhance Population Viability of Endangered Birds. *PLoS ONE* 3: e4084.

Ortega, E., Mañosa, S., Margalida., Sánchez, R., Oria, J. and González, L.M. (2009). A demographic description of the recovery of the Vulnerable Spanish imperial eagle *Aquila adalberti*. *Oryx* 43: 113-121

Palomares, F., Rodríguez, A., Revilla, E., López-Bao, J. V. and Calzada, J. (2011). Assessment of the Conservation Efforts to Prevent Extinction of the Iberian Lynx. *Conservation Biology* 25: 4-8.

Pullin, A. S., Knight, T. M., Stone, D. A. and Charman, K. (2004). Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation* 119: 245-252.

Pullin, A. S., and Knight, T. M. (2005). Assessing Conservation Management's Evidence Base: a Survey of Management-Plan Compilers in the United Kingdom and Australia. *Conservation Biology* 19: 1989-1996.

Pullin, A. S. and Stewart, GB. (2006). Guidelines for systematic review in conservation and environmental management. *Conservation Biology* 20:1647–1656.

Salafsky, N., Margoulis, R. and Redford, K. (2001). *Adaptive management: A tool for conservation practitioners*. Biodiversity Support Program. Washington, D.C.

Sánchez, J. J. (1998). The recovery of the Black Vulture *Aegypius monachus* in Spain. In: Tewes, E., Sánchez, J. J., Heredia, B. and Bijleveld van Lexmond, D. (Eds.) *Proceedings of the International Symposium on the Black Vulture in SE Europe and adjacent regions*, pp. 89-99. Black Vulture Conservation Foundation-Frankfurt Zoological Society. Palma de Mallorca.

Sánchez, R., Margalida, A., González, L.M. and Oria, J. (2009). Temporal and spatial differences in the feeding ecology of the Spanish Imperial Eagle *Aquila adalberti* during the non-breeding season: effects of the rabbit population crash. *Acta Ornithologica* 44: 53-58

Sekercioglu, C. H., Daily, G. C. and Ehrlich, P. R. (2004) Ecosystem consequences of bird declines. *Proceedings of the National Academy of Sciences USA* 101: 18042–18047.

Sokal, R. R. and Rohlf, F. J. (1995). *Biometry: the principles and practice of statistics in biological research*. 3rd edition. W. H. Freeman and Co. New York.

Soulé, M. E. (1986). *Conservation Biology: the science of scarcity and diversity*. Sinauer Associates, Inc. Sunderland.

Suetens, V. and Van Groenendal, P. (1966). Sobre ecología y conducta reproductiva del buitre negro (*Aegypius monachus*). *Ardeola* 12: 19-44.

StatSoft (2002). *Statistica 6.1*. Tulsa. www.statsoft.com. Accessed 11 March 2012

Tewes, E. (1996). *The European Black Vulture Aegypius monachus: management techniques and habitat requirements*. PhD Thesis. University of Vienna. Vienna.

Valverde, J. A. (1966). Sobre buitres negros en Andalucía. *Ardeola* 12: 101-115.

Zhang, L., Wang, M. H., Hu, J. and Ho, Y. S. (2010). A review of published wetland research, 1991-2008: ecological engineering and ecosystem restoration. *Ecological Engineering* 36: 973-980.

Zuberogoitia, I., Martínez, J. E., Margalida, A., Gómez, I., Azkona, A and Martínez, J. A. (2010) Reduced food availability induces behavioural changes in griffon vulture. *Ornis Fennica* 87: 52-60.

Appendix S1. Publications on Cinereous Vulture *Aegypius monachus* in España, integrated in the analysis of the present article. It is specified the author/s, the name of the publication, the year of releasing, the type of publication (SCI journals, other technical-scientific articles in periodic journals, legal-divulgation, books-PhD thesis and chapters-articles in abstracts of monographs, workshops, congresses or meetings) and the subject treated (biology-ecology, conservation-threats-management, census and others).

Author/s	Name of the publication	year	type of publication	subject
Bernis	*Ardeola*	1966	other scientific-technical articles	others
Suetens & Groenendal	*Ardeola*	1966	other scientific-technical articles	biology-ecology
Valverde	*Ardeola*	1966	other scientific-technical articles	others
Garzón	*Ardeola*	1968	other scientific-technical articles	others
Elosegui	*Munibe*	1970	other scientific-technical articles	census
Elosegui	*Ardeola*	1971	other scientific-technical articles	census
Morillo & Lalanda	*Bol. Est. Central Ecol.*	1972	other scientific-technical articles	biology-ecology
Mayol	*Aegypius*	1973	other scientific-technical articles	census
Suárez	*Ardeola*	1973	other scientific-technical articles	biology-ecology
Bernis	*Ardeola*	1974	other scientific-technical articles	biology-ecology
Garzón	*Ardeola*	1974	other scientific-technical articles	others
Hiraldo	*Naturalia Hispanica*	1974	other scientific-technical articles	census
Mayol	*Ardeola*	1975	other scientific-technical articles	conservation-threats-management
Richford et al.	*Ardeola*	1975	other scientific-technical articles	census
Alvarez et al.	*Ornis Scandinavica*	1976	SCI journals	biology-ecology
Hiraldo	*Doñana Acta Vertebrata*	1976	other scientific-technical articles	biology-ecology
Mayol	*Aegypius*	1976	other scientific-technical articles	census
Amores	*PhD Thesis*	1977	books-PhD Thesis	others
Amores et al.	*Book Raptors of Sierra Morena*	1977	books-PhD Thesis	others
Hiraldo	*PhD Thesis*	1977	books-PhD Thesis	others
Acedo & Ruiz	*Oxyura*	1978	other scientific-technical articles	census
Mayol	*Bol. Soc. Hist. Nat. Bal.*	1978	other scientific-technical articles	others
Torres et al.	*Bolet. Est. Centr. Ecol.*	1980	other scientific-technical articles	others
Hiraldo	*Naturalia Hispanica*	1981	other scientific-technical articles	biology-ecology
Mayol	*Quercus*	1982	legal-divulgation	others
Hiraldo	*Vulture Biology Manag.*	1983	other scientific-technical articles	biology-ecology
De Juana y De Juana	*Rapinyaires mediterranis*	1984	other scientific-technical articles	biology-ecology
Martín-Mateo et al.	*EOS*	1984	SCI journals	biology-ecology
Caballero	*Panda*	1986	legal-divulgation	census
González et al.	*Congress Mediterranean Raptors*	1986	chapters-articles in abstracts	census
Hernández et al.	*J. Field Ornithology*	1986	SCI journals	conservation-threats-management
Muntaner & Tewes	*La Garcilla*	1989	legal-divulgation	others
Bermejo	*Congress CODA*	1990	chapters-articles in abstracts	census
González	*Quercus*	1990	legal-divulgation	census
Hiraldo & Donázar	*Bird Study*	1990	SCI journals	biology-ecology

Jiménez	*Book imperial eagle/black vulture Castilla-La Macha*	1990	books-PhD Thesis	biology-ecology
Jiménez	*Congress CODA*	1990	chapters-articles in abstracts	biology-ecology
Jiménez	*Congress CODA*	1990	chapters-articles in abstracts	biology-ecology
Ruiz et al.	*Congress CODA*	1990	chapters-articles in abstracts	biology-ecology
Ruiz Romero et al.	*Testudo*	1990	other scientific-technical articles	census
Tewes	*Quercus*	1991	legal-divulgation	census
Viada	*Boleta*	1991	other scientific-technical articles	conservation-threats-management
Adecab	*Quercus*	1992	legal-divulgation	conservation-threats-management
Blanco & González	*Red Book*	1992	other scientific-technical articles	conservation-threats-management
Sánchez et al.	*Aegypius*	1992	other scientific-technical articles	census
Donázar	*Book Los buitres ibéricos*	1993	books-PhD Thesis	others
Tewes	*Raptor conservation today*	1994	other scientific-technical articles	conservation-threats-management
Alvarez & Garcés	*Holarctic birds of prey*	1995	chapters-articles in abstracts	biology-ecology
Alvarez & Garcés	*Congress Mediterranean Raptors*	1996	chapters-articles in abstracts	biology-ecology
Andrés et al	*Congress Mediterranean Raptors*	1996	chapters-articles in abstracts	census
Galán et al.	*Ecología*	1996	other scientific-technical articles	conservation-threats-management
Heredia	*European Action Plan*	1996	legal-divulgation	conservation-threats-management
Tewes	*PhD Thesis*	1996	books-PhD Thesis	others
Torres et al.	*Congress Mediterranean Raptors*	1996	chapters-articles in abstracts	census
Blanco et al.	*J. Raptor Research*	1997	SCI journals	biology-ecology
Galán	*Quercus*	1997	legal-divulgation	conservation-threats-management
Galán et al.	*Congress on scavenger raptors*	1997	chapters-articles in abstracts	biology-ecology
De Andrés et al.	*Congress Cinereous Vulture*	1998	chapters-articles in abstracts	census
Fargallo et al	*J. Raptor Research*	1998	SCI journals	biology-ecology & conservation
Galán et al.	*Holarctic birds of prey*	1998	chapters-articles in abstracts	conservation-threats-management
Galán et al.	*Holarctic birds of prey*	1998	chapters-articles in abstracts	conservation-threats-management
Gentil & Ventanas	*Holarctic birds of prey*	1998	chapters-articles in abstracts	others
Guzmán & Jiménez	*Holarctic birds of prey*	1998	chapters-articles in abstracts	biology-ecology
Sánchez	*Congress Cinereous Vulture*	1998	chapters-articles in abstracts	census
Wink et al. 1998	*Molecular Ecology*	1998	SCI journals	biology-ecology
Martí et al.	*Workshop P. N. Peñalara*	1999	chapters-articles in abstracts	census

Oria	*Vulture News*	1999	other scientific-technical articles	biology-ecology
Atienza et al	*Raptor Research Foundation Congress*	2001	chapters-articles in abstracts	biology-ecology
Corbacho et al.	*Raptor Research Foundation Congress*	2001	chapters-articles in abstracts	biology-ecology
Costillo et al.	*Congress SPAs Extremadura*	2001	chapters-articles in abstracts	census
Costillo et al.	*Raptor Research Foundation Congress*	2001	chapters-articles in abstracts	conservation-threats-management
Moleón et al.	*Raptor Research Foundation Congress*	2001	chapters-articles in abstracts	biology-ecology
Sánchez	*Congress on poisoning*	2001	chapters-articles in abstracts	conservation-threats-management
Blanco et al.	*Workshop P. N. Peñalara*	2002	chapters-articles in abstracts	conservation-threats-management
Costillo et al.	*Workshop P. N. Peñalara*	2002	chapters-articles in abstracts	biology-ecology
De la Puente et al	*Quercus*	2002	legal-divulgation	others
Del Moral et al.	*Workshop P. N. Peñalara*	2002	chapters-articles in abstracts	census
Donázar	*Workshop P. N. Peñalara*	2002	chapters-articles in abstracts	others
Donázar et al.	*Ecological Applications*	2002	SCI journals	biology-ecology & conservation
Galán	*Andalus*	2002	other scientific-technical articles	census
Godino et al.	*Vulture News*	2002	other scientific-technical articles	census
Jiménez	*Anuario Ornitologico Ciudad Real*	2002	other scientific-technical articles	biology-ecology
Margalida & Heredia	*Sandgrouse*	2002	other scientific-technical articles	biology-ecology
Ministerio Medio Ambiente	*Royal Decree 1098/2002*	2002	legal-divulgation	conservation-threats-management
Muñoz	*Atlas wintering birds Madrid*	2002	other scientific-technical articles	others
Villegas et al.	*Comparative Biochemistry*	2002	SCI journals	biology-ecology
Galán et al.	*Quercus*	2003	legal-divulgation	census
Junta Castilla-La Mancha	*Conservation Plan*	2003	legal-divulgation	conservation-threats-management
Sánchez	*Atlas breeding birds Spain*	2003	other scientific-technical articles	others
Tewes	*Anuario Ornit. Baleares*	2003	other scientific-technical articles	conservation-threats-management
Arenas	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Azcárate & Carbonell	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Caldera	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Costillo	*PhD Thesis*	2004	books-PhD Thesis	biology-ecology & conservation

Costillo et al.	*Spanish Ornithological Congress*	2004	chapters-articles in abstracts	biology-ecology
Costillo et al.	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	biology-ecology
De la Puente	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
De la Puente	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	others
De la Puente	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	biology-ecology
Dobado	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Galán & Segovia	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	conservation-threats-management
Godino et al.	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Luque et al.	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Martín et al.	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Martínez et al.	*J. Parasithology*	2004	SCI journals	biology-ecology
Mayol	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	others
Moreno-Opo et al.	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	conservation-threats-management
Moreno-Opo et al.	*Trofeo*	2004	legal-divulgation	conservation-threats-management
Mosqueda	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Sánchez	*Red Book*	2004	other scientific-technical articles	conservation-threats-management
Tewes	*Anuario Ornit. Baleares*	2004	other scientific-technical articles	census
Tewes	*Raptors Worldwide*	2004	chapters-articles in abstracts	conservation-threats-management
Tewes	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	conservation-threats-management
Vielva	*Symposium Cinereous Vulture Cordoba*	2004	chapters-articles in abstracts	census
Villegas et al.	*J. Raptor Research*	2004	SCI journals	biology-ecology
Cano & Hernández	*Quercus*	2005	legal-divulgation	conservation-threats-management
Carrete y Donázar	*Biological Conservation*	2005	SCI journals	biology-ecology
Cuevas y De la Puente	*Unedited report*	2005	other scientific-technical articles	biology-ecology
De la Puente y Gamonal	*Gypas Conference*	2005	chapters-articles in abstracts	census
Jiménez	*Gypas Conference*	2005	chapters-articles in abstracts	others
Junta Extremadura	*Conservation Plan*	2005	legal-divulgation	conservation-threats-management
SEO-Salamanca	*Quercus*	2005	legal-divulgation	conservation-threats-management

Blanco	*Quercus*	2006	legal-divulgation	conservation-threats-management
De la Puente	*Gypas Conference*	2006	chapters-articles in abstracts	census
Dobado y Arenas	*Quercus*	2006	legal-divulgation	conservation-threats-management
Jiménez	*Gypas Conference*	2006	chapters-articles in abstracts	biology-ecology
Morán-López et al	*Animal Conservation*	2006	SCI journals	biology-ecology
Morán-López et al	*Biological Conservation*	2006	SCI journals	biology-ecology
Morán-López et al	*Book Conservation Extremadura*	2006	other scientific-technical articles	biology-ecology
Costillo et al.	*Ardea*	2007	SCI journals	biology-ecology & conservation
Costillo et al.	*Ardeola*	2007	SCI journals	biology-ecology & conservation
De la Puente et al	*National census 2006*	2007	books-PhD Thesis	census
Ministerio Medio Ambiente	*Royal Decree 664/2007*	2007	legal-divulgation	conservation-threats-management
Moreno-Opo & Guil	*Handbook management*	2007	books-PhD Thesis	others
De la Puente et al	*Congress scavenger raptors*	2008	chapters-articles in abstracts	conservation-threats-management
González & Moreno-Opo	*Ambienta*	2008	legal-divulgation	conservation-threats-management
Hernández y Margalida	*Ecotoxicology*	2008	SCI journals	conservation-threats-management
Lemus et al.	*Plos ONE*	2008	SCI journals	conservation-threats-management
Moreno-Opo et al.	*Jara y Sedal*	2008	legal-divulgation	conservation-threats-management
Poulakakis et al.	*Biol. J. Linnean Society*	2008	SCI journals	biology-ecology
WWF-Adena	*Poisoning report*	2008	legal-divulgation	conservation-threats-management
Blanco et al.	*Animal Conservation*	2009	SCI journals	conservation-threats-management
Donázar et al.	*Science*	2009	SCI journals	conservation-threats-management
Garcia y Moreno-Opo	*Munibe*	2009	other scientific-technical articles	conservation-threats-management
Lemus & Blanco	*Proceedings RSBC*	2009	SCI journals	conservation-threats-management
Moreno-Opo et al.	*Munibe*	2009	other scientific-technical articles	biology-ecology
Moreno-Opo et al.	*Wildlife Biology*	2009	SCI journals	biology-ecology
Rodríguez-Ramos et al	*Congress on lead intoxication*	2009	chapters-articles in abstracts	conservation-threats-management
De la Puente	*Spanish Ornithological Congress*	2010	chapters-articles in abstracts	biology-ecology
Del Moral & De la Puente	*Vertebrados ibéricos*	2010	legal-divulgation	others
Margalida et al.	*J. Applied Ecology*	2010	SCI journals	conservation-threats-management
Moreno-Opo et al.	*Ardeola*	2010	SCI journals	biology-ecology

Álvarez et al.	*Book forest-dwelling raptors*	2011	other scientific-technical articles	conservation-threats-management
De la Puente et al	*Book forest-dwelling raptors*	2011	other scientific-technical articles	biology-ecology
Donázar et al.	*Book forest-dwelling raptors*	2011	other scientific-technical articles	conservation-threats-management
Jiménez y González	*Ecología*	2011	other scientific-technical articles	biology-ecology
Junta Andalucía	*Conservation Plan*	2011	legal-divulgation	conservation-threats-management
Margalida & Moreno-Opo	*Animal Conservation*	2011	SCI journals	conservation-threats-management
Margalida et al.	*Nature*	2011	SCI journals	conservation-threats-management
Margalida et al.	*Science*	2011	SCI journals	conservation-threats-management
Ministerio Medio Ambiente	*Royal Decree 1632/2011*	2011	legal-divulgation	conservation-threats-management
Morán-López et al	*Book forest-dwelling raptors*	2011	other scientific-technical articles	biology-ecology & conservation
Moreno-Opo et al.	*Biodiv Cons*	2011	SCI journals	conservation-threats-management
Moreno-Opo et al.	*Plos ONE*	2011	SCI journals	biology-ecology & conservation

DISCUSIÓN

Efecto de la metodología y de factores ecológicos sobre la selección de hábitat de nidificación

La reproducción de los animales silvestres está condicionada por los procesos naturales y/o actividades humanas que acontecen en el medio natural, y su estudio también está relacionado con el tipo de metodología empleada (Katzner et al. 2007, Magana et al. 2010). Existen pocos casos en los que se han podido extraer conclusiones de los factores que determinan la elección de los lugares de reproducción para una especie a distinta escala espacial y en localidades geográficas diferentes. En este sentido, los metanálisis pueden constituir una buena herramienta para evaluar e integrar los resultados de estudios previos, exponiendo conclusiones sobre los factores ecológicos y antrópicos que afectan a hábitats y especies objeto de estudio (Pullin y Stewart 2006). En este tipo de análisis es importante contemplar y soslayar algunas dificultades que presenta la información proveniente de los distintos estudios a comparar, como son las deficiencias en la toma de datos en distintos trabajos, la falta de criterios homogéneos en la recogida de datos y la variación en los requerimientos ecológicos de la especie objeto de modelización entre distintas áreas (Hoeksema y Forde 2008, Benítez-López et al. 2010)

Distintos estudios han mostrado patrones comunes de selección de hábitat de nididificación del buitre negro (Fargallo et al. 1998, Donázar et al. 2002, Jiménez 2002, Poizaridis et al. 2004, Cuevas y De la Puente 2005, Morán-López et al. 2005, Gavashelishvili et al. 2006). A pesar de estas conclusiones, existieron diferencias en relación a las variables que resultaron estadísticamente significativas entre los estudios así como divergencias en los métodos experimentales aplicados, lo que pudo condicionar los resultados finales. Así, en el Capítulo 1 de esta Tesis Doctoral se evaluó, a través de un metanálisis, cómo los procedimientos metodológicos influyeron en la variabilidad recogida en los resultados de los distintos estudios y, en consecuencia, cómo afectó a la selección del hábitat de nidificación del buitre negro a escala global (Biostat 2006). Los metanálisis son técnicas estadísticas que combinan datos de varios estudios con el propósito de identificar y cuantificar comparativamente los factores que determinan la similitud y variación entre los resultados de dichos estudios (Biostat 2006).

El análisis conjunto de los trabajos previos de selección de hábitat y del trabajo de campo presente mostró que el buitre negro selecciona, para ubicar sus nidos, grandes árboles en laderas pronunciadas, en el tercio superior de éstas, y orientadas principalmente hacia el Sur y Este, en la proximidad de canchales y alejado de infraestructuras humanas o de factores antropogénicos que pudieran provocar molestias. Las laderas con mayor pendiente pudieron resultar seleccionadas por su mayor inaccesibilidad y lejanía al hombre y, en consecuencia, por su mayor tranquilidad ante molestias (González et al. 2006a, Aubad et al. 2010). Por otro lado, los árboles más grandes soportan mejor el peso de grandes nidos y, al ser más altos que el resto, permiten una mayor facilidad de aterrizaje y despegue de un ave como el buitre negro (Fargallo et al. 1998, Donázar et al. 2002). Esta mayor robustez de los árboles se localiza a menudo alrededor de canchales y pedrizas (Thompson 2005) lo que, además de facilitar el acceso a la plataforma de nidificación, permite una mayor detectabilidad de predadores y de otro tipo de perturbaciones (Blumstein et al. 2005, Margalida et al. 2011c). Las laderas orientadas hacia el Este y Sur proporcionarían unas mejores condiciones para el establecimiento del nido, tanto por la presencia de árboles grandes respecto al resto de dosel arbóreo de menor tamaño como por unas mejores condiciones climatológicas a escala local (Pérez 2005).

En relación al análisis de la influencia de la metodología empleada, el número total de variables estudiadas influyó en los resultados de los modelos y en la variabilidad recogida por éstos (Fielding y Bell 1997, Drew et al. 2010). Ello supone que en los estudios de modelización del hábitat, tanto el número como el tipo de variables deben ser elegidos cuidadosamente a priori (Guisan y Thuiller 2005, Jiménez-Valverde et al. 2008). Por ello, resulta especialmente valiosa la elección de variables lo más explicativas y repetibles entre estudios y localidades (Katzner et al. 2007). Además, es importante considerar la influencia que pueden tener otro tipo de factores además de los habitualmente empleados en este tipo de estudios (vegetación, climatología, geomorfología e influencia antrópica), y que pudieran incorporarse a estudios de modelización. Entre ellos se hallan cuestiones relacionadas con el cambio global, los procesos ecológicos o los factores bioquímicos (Allen et al. 2002, Hijmans y Graham 2006). El emplazamiento de los puntos aleatorios de muestreo determina la variabilidad incluida en el estudio. Así, si los puntos aleatorios se sitúan dentro de los límites de la colonia, la información resultante será más detallada en cuanto a los factores que operan a escala local (Fielding y Bell 1997, Hirzel y Guisan 2005, Morán-López et al. 2005). Por otro lado, los resultados del metanálisis mostraron que si los datos se toman en un solo año la variabilidad explicada por el modelo se incrementa (Scott et al. 2002). A diferencia de esta conclusión, otros estudios han mostrado que los sesgos en los resultados de trabajos de modelización se reducen y la variabilidad recogida aumenta cuando los datos se toman a lo largo de un período temporal prolongado, debido a la atenuación y absorción de los efectos que pudieran operar de manera particular durante un único año (Drew et al. 2010, Brotons et al. 2010). La elección de puntos como unidad de muestreo aleatorio permitió englobar una mayor devianza/varianza en el modelo respecto al uso de polígonos. Ello pudo ser causado por la mayor precisión que ofrecen los datos obtenidos en un punto concreto, acerca de cuestiones relacionadas con la selección ejercida por una especie (Donázar et al. 2002, Segurado y Araujo 2004). Por otro lado, la proporción de nidos estudiados respecto a muestras aleatorias debe ser lo más equilibrada posible, de manera que cuando esta relación se aproxima a 1 o incluso es favorable a las muestras aleatorias, la variabilidad explicada por el modelo se incrementa (Fielding y Bell 1997, Scott et al. 2002, Engler et al. 2004).

Los resultados también indicaron que el tipo de hábitat y el tamaño de la colonia determinan el tipo de variable más influyente sobre la selección del emplazamiento de nidificación. En este sentido, las actividades humanas tuvieron mayor influencia sobre las colonias situadas en bosques mediterráneos de *Quercus*. Es posible que los buitres negros hayan tenido una mayor relación con las actividades humanas en hábitats como los pinares, que han sido objeto de un mayor aprovechamiento secular, en muchos casos sostenible con los requerimientos de la especie (Lindenmayer et al. 2000, Paillet et al. 2009). En cambio, en los bosques de vegetación típica mediterranea solo se desarrollan actividades económicas de manera ocasional y, en muchos casos, durante la época de cría (descorche, repaso caminos, etc), lo que ha podido provocar una mayor sensibilidad del buitre negro hacia la presencia del hombre (Blumstein et al. 2005).

Entre las conclusiones emanadas de este Capítulo I cabe reseñar la importancia y valor que tiene el reconocimiento de la selección de hábitat de una especie como es el buitre negro, para optimizar las acciones de conservación en base a evidencias de conocimiento, tanto para la propia especie como para las que habitan el mismo medio (Sutherland et al. 2004, Branton y Richardson 2011, Olea y Mateo-Tomás 2011). En este sentido, la modelización de los requerimientos de las especies contribuye a mejorar tanto las acciones de conservación ex situ como in situ. Así, ambos tipos de programas deberían tener en cuenta los patrones globales de selección de hábitat, así como los análisis de viabilidad poblacional que se realicen al respecto (Armstrong y Seddon 2008, Sutherland et al. 2010).

Factores ambientales que determinan el éxito reproductivo

El estudio y conocimiento en detalle de las estrategias de vida de las especies resulta esencial para la adopción de medidas de conservación, sobre todo cuando sus poblaciones están amenazadas (Pullin et al. 2004, Oro et al. 2008). Para desarrollar dicho conocimiento distintas variables poblacionales resultan muy reveladoras. Entre ellas, el éxito reproductivo es un parámetro que depende de un amplio número de factores como son, por ejemplo, las condiciones climatológicas (Kostrzewa y Kostrzewa 1991, Margalida et al. 2007), las características del hábitat, la disponibilidad de alimento, la capacidad para eludir a depredadores (Lind y Creswell 2005, Wilkin et al. 2009) y las molestias de origen humano (Zuberogoitia et al. 2008, Margalida et al. 2011c). La puesta en marcha de programas de recuperación de especies amenazadas requiere la comprensión tanto de la selección de hábitat ejercida como de los factores que determinan el éxito reproductivo (Lindenmayer et al. 2008). En este Capítulo 2 se evaluó la influencia de variables ambientales y de actividades humanas sobre el éxito reproductivo del buitre negro, con el propósito de identificar cuáles de ellas son más incidentes y, en consecuencia, las que habrían de ser tenidos en cuenta para la gestión de las áreas habitadas por la especie.

Los resultados obtenidos mostraron cómo variables relativas al estado de conservación del nido, la altura del árbol-nido, el diámetro del nido y la vegetación a escala de paisaje (mayor cobertura de árboles, arbustos y pedrizas, y mayor altura de matorral) tuvieron repercusión significativa positiva sobre el éxito reproductivo del buitre negro. En primer lugar, para llevar a cabo la reproducción con éxito es necesario que la estructura del nido sea la adecuada y no se produzcan problemas de falta de sustentabilidad o acondicionamiento (Margalida y Bertran 2000, Dalley et al. 2008). Por ello, los buitres realizan aportes continuados de material al nido, para asegurar la estructura y tapizado del mismo. Por otro lado, los árboles más altos facilitan las operaciones necesarias para un adecuado proceso de cría: es más fácil acceder a los nidos situados en dichos árboles (Donázar et al. 2002), aumenta la probabilidad de detección de predadores u otras fuentes de molestias (Magana et al. 2010, Margalida et al. 2011c) y soportan de manera más adecuada el peso de nidos tan grandes como los del buitre negro (Fargallo et al. 1998). En relación a la vegetación, las variables resultantes significativas podrían tener relación con aspectos como la accesibilidad al lugar de cría, de modo que bosques densos reducen la posibilidad de llegar a las inmediaciones del nido por parte de predadores terrestres y personas (Aubad et al. 2010). También podrían relacionarse con la selección positiva de determinadas especies, como son los alcornoques o los enebros (datos propios).

El Capítulo 2 no evidenció influencia significativa de molestias humanas sobre el éxito reproductivo. No obstante, este tipo de variables sí tuvieron efecto en las colonias de cría estudiadas en estudios previos (Donázar et al. 2002). En ese caso, se halló que una menor presencia humana tuvo efectos positivos sobre el éxito reproductivo tal y como ocurre para la selección del emplazamiento de los nidos (Fargallo et al. 1998, Poizaridis et al. 2004, Morán-López et al. 2005, Gavashelishvili et al. 2006). Sobre las variables a escala de paisaje, además, los estudios previos indicaron unas mayores tasas reproductivas en zonas de mayor altitud relativa, en laderas escarpadas, pronunciadas y no orientadas al norte (Donázar et al. 2002, Morán-López et al. 2006). Una cuestión importante que podría condicionar el éxito reproductivo del buitre negro que no ha sido tenida en cuenta en los estudios, incluido el presente, es la propia calidad individual de las aves (Robinson et al. 2005, Sanz-Aguilar et al. 2008). En distintas especies de aves existe una correlación entre las características físicas del individuo, la edad o experiencia y la calidad del lugar de reproducción (véase por ejemplo Velando y Freire 2001). Así, individuos en mejor estado físico y mejores capacidades competitivas seleccionarían determinadas características de los hábitat que estarían relacionadas con la calidad general del área de cría (Lascroel et al. 2009). La inclusión de estos factores en el presente capítulo podría haber incrementado la precisión y representatividad de los resultados, aunque no se llevó a cabo por la dificultad de evaluar la calidad individual de las aves de la población objeto de estudio.

De forma general, los bosques maduros que albergan árboles robustos y altos deben ser mantenidos de manera prioritaria. Estas masas boscosas constituyen áreas fuente para la reproducción contribuyendo a la producción continuada de jóvenes y, en consecuencia, a mejorar el estado de conservación de la especie (Lindenmayer et al. 2000). Además, estas áreas forestales maduras en laderas pronunciadas son también el habitat de reproducción de otras grandes rapaces. Además de proteger el hábitat de cría, no se debe perder de vista los aspectos que condicionan en mayor medida la conservación de una especie de rapaz longeva: la necesidad de garantizar una adecuada supervivencia adulta mediante la lucha contra el uso ilegal de cebos envenenados y accidentes contra infraestructuras humanas (Oro et al. 2008, Carrete et al. 2009a, Ortega et al. 2009), y el mantenimiento de superficies mínimas de hábitat en las que satisfacer los requerimientos de alimentación y descanso (Harris et al. 2005).

Área de campeo y alimentación

La abundancia relativa, distribución y tamaño de las presas influyen sobre la dimensión y características de las áreas de campeo de las rapaces necrófagas (Costillo et al. 2007c, Margalida et al. 2009) y, en consecuencia, determinan el comportamiento alimentario y la territorialidad asociada (Hiraldo 1977, Donázar 1993). Las áreas de alimentación del buitre negro han sido estudiadas en distintas regiones europeas (Carrete y Donázar 2005, Costillo 2005, Vasilakis et al. 2006), habiéndose dirigido dichos trabajos a conocer su extensión y las posibles variaciones existentes entre clases de individuos y períodos. Esta base de conocimiento es clave para analizar la disponibilidad de alimento existente, los patrones de selección de hábitat o las amenazas que se ciernen sobre la especie.

La principal particularidad del trabajo del Capítulo 3 fue el empleo de un método de recopilación de información original, económico y novedoso, como es la recogida de crotales de ganado, ingeridos por los buitres negros junto con la carroña y posteriormente expulsados en egagrópilas, y el hallazgo posterior de la explotación ganadera de procedencia del crotal. Este método de asignación de localidades de alimentación es distinto al empleado en los trabajos previos (Carrete y Donázar 2005, Costillo 2005, Vasilakis et al. 2006), que usaron la radiotelemetría de determinados individuos. El marcaje con emisores de radio y la recepción de las señales permite obtener un elevado número de localizaciones de cada ejemplar. De acuerdo con las características del ave marcada, se pueden evaluar variaciones de los movimientos y territorios ocupados según la edad y sexo, así como diferencias en función de la época del año. Además, las localizaciones con radiotelemetría informan sobre el uso general del espacio para la obtención de cualquier presa potencial de la especie (Corbacho et al. 2007), a diferencia del empleo de crotales que solo informa sobre la ingestión de alimento de origen ganadero (que son los que obligatoriamente han de estar identificados con crotales). No obstante, el uso de crotales como técnica de localización tiene distintas ventajas metodológicas. La muestra de aves para las que se obtienen patrones de alimentación es mucho mayor que con individuos radiomarcados; con esta técnica se informa acerca del conjunto de aves que regurgitaron crotales en la base de 88 nidos distintos y en 11 posaderos de la colonia, a lo largo de cinco años de muestreo. Por el contrario, en los trabajos de radioseguimiento, el máximo número de buitres negros marcados fue 14 (Carrete y Donázar 2005). Además, el hallazgo de la procedencia de los crotales informa sobre una localidad inequívoca de alimentación, a escala de explotación ganadera, a diferencia del radioseguimiento donde resulta muy complejo discernir la actividad concreta realizada por el ave, en este caso la ingestión de alimento, en cada punto del territorio.

La distancia recorrida para la adquisición del alimento, tomando como punto de partida la colonia de cría, resultó mayor en promedio que en otros estudios (26,3 ± 36,1 km, respecto a los 16,1 ± 14,0 km expuestos en Corbacho et al. 2010). Es destacable que más de la mitad de las localizaciones de alimentación (54,5%) procedieron de un único punto, un muladar con aportes continuados de carroña de aves necrófagas. Este hecho ilustra la importancia de este tipo de lugares para la gestión de la alimentación de aves necrófagas (Donázar 1992, Donázar et al. 2009a, 2009b), así como la probable alteración de sus patrones naturales de campeo y alimentación, que han sido dirigidos de manera rutinaria hacia una localidad donde el alimento resulta abundante y predecible (Robb et al., 2008; Carrete et al. 2009b).

La conservación del buitre negro depende, en parte, de la gestión del hábitat donde se encuentra. Mientras que las áreas de cría han sido objeto de una elevada cobertura de protección oficial (Moreno-Opo y Guil 2007), los esfuerzos de protección de las áreas de campeo y alimentación son todavía incipientes (Carrete y Donázar 2005). Las principales amenazas a las que se enfrentan ésta y otras rapaces necrófagas provienen de la escasez, alteración y escasa calidad del alimento (González y Moreno-Opo 2008, Hernández y Margalida 2008, Margalida et al. 2008b, Hernández y Margalida 2009b). Por ello, una de las medidas de conservación más eficaces proviene de la necesaria dotación de recursos tróficos, con el fin de asegurar su disponibilidad y calidad (Donázar et al. 2009b). El primer paso para ejecutar dichas medidas es conocer y delimitar las zonas donde la especie se encuentra presente en los distintos períodos de su ciclo vital.

Presencia de buitres negros en carroñas

Las principales presas que conforman la dieta de los buitres en Europa son los cadáveres de ganado extensivo y de distintas especies silvestres, sobre todo ungulados y lagomorfos (ver revisión en Donázar et al. 2009b). Estas presas las obtienen tanto del medio natural como de muladares, entendidos éstos como los lugares donde se depositaban los cadáveres de ganado doméstico de forma tradicional y, más recientemente, de puntos de alimentación suplementaria. Estos últimos han sido implantados con el propósito de intentar incrementar las poblaciones de buitres y sus parámetros reproductivos, facilitar la expansión geográfica de las especies y reducir los riesgos derivados del consumo de presas contaminadas (Donázar et al. 2009b, Margalida et al. 2010).

Una adecuada gestión de los recursos tróficos resulta determinante para la conservación de las especies amenazadas (BirdLife International 2004, Jones 2004). En el caso de rapaces necrófagas, el conocimiento detallado de la dieta y de las preferencias por determinadas partes de las carroñas, puede constituir una herramienta clave para el diseño de estrategias de conservación (Margalida et al. 2009). En este Capítulo 3 se analizó el uso de carroñas y los patrones de selección de distintos formatos de éstas por parte del buitre negro, con el objetivo de proporcionar recomendaciones de actuación para el establecimiento de puntos de alimentación suplementaria.

Distintas variables explicaron una presencia más numerosa de buitres negros en las carroñas y una optimización de ésta. En primer lugar, la cantidad de biomasa proporcionada se correlacionó positivamente con el número total de buitres negros que acudieron a la carroña e inversamente con el tiempo transcurrido desde el depósito de ésta hasta el inicio del consumo. El tiempo promedio que emplearon los buitres negros en acudir a las carroñas (214 min) resultó menor que el hallado para la misma especie en el Cáucaso (1122 min, Gavashelishvili y McGrady 2006). Ello pudo deberse a la proximidad de núcleos poblacionales y a una alta abundancia relativa de individuos en el área de estudio. En este sentido, el elevado promedio de individuos observados por carroña (23.4 ejemplares) sugiere la comunicación entre congéneres para la localización del alimento, lo que probablemente mejora el éxito de campeo y alimentación (Jackson et al. 2008). Por otro lado, el formato en que se aporta la carroña determinó tanto el tiempo que tardaron en acceder como el número total de buitres negros que acudieron a alimentarse. Así, cuando se proporcionaron restos dispersos, los buitres negros comenzaron antes a alimentarse que cuando se depositaron en un mismo punto, lo que favoreció sobre todo al buitre leonado. Respecto a las preferencias por distintas partes de la carroña, el buitre negro seleccionó principalmente restos pequeños o medianos, de partes cárnicas duras o relativamente duras (músculo, tendón, piel, etc), preferiblemente cuando se encontraban dispersos y no concentrados en un único punto (König 1983). De este modo el buitre negro podría verse favorecido por el consumo de los mencionados formatos (Hertel 1994). En resumen, el buitre negro se benefició de carroñas suministradas en grandes cantidades de biomasa, despiezadas en trozos pequeños y medianos, dispersos y no concentrados en un único punto. Del mismo modo, si el número de piezas en que se divide el aporte es elevado se optimizará la presencia de la especie.

La gestión adecuada de la alimentación es uno de los grandes retos a los que se enfrentan gestores y especialistas en conservación de aves necrófagas. En especies amenazas, la alimentación suplementaria resulta muy útil para aumentar las tasas de supervivencia preadultas (Oro et al. 2008) y de parámetros reproductivos (González et al. 2006b, López-Bao et al. 2008). Los muladares bien gestionados, además, contribuyen a reducir la mortalidad no natural mediante el aporte de alimento no contaminado (Gilbert et al. 2007, Margalida et al. 2008b). Sin embargo, la alimentación suplementaria también puede tener repercursiones negativas sobre la dinámica poblacional (p. ej. Robb et al. 2008), por lo que, de acuerdo con la coevolución entre buitres y sus presas, y tal y como ha ocurrido durante los últimos siglos, la alimentación de las especies necrófagas debe estar basada principalmente en el aprovechamiento sostenible de cadáveres de ganado extensivo o de animales silvestres, que se presentan en el territorio de manera espacial y temporalmente heterogénea y pulsada (Donázar et al. 2009b, Cortés-Avizanda 2010). No obstante, cuando se plantee la puesta en marcha de puntos de alimentación suplementaria para buitres negros, los gestores deberían tener en cuenta variables relativas a cantidad, formato, número y dispersión de trozos en que se aporta la carroña, con el propósito de optimizar su consumo.

Conservación de especies amenazadas y aprovechamientos económicos: descorche en colonias de buitre negro

Las relaciones entre actividades humanas y especies animales silvestres es un asunto de preocupación e interés para gestores y legisladores, que buscan soluciones que permitan hacer compatible la conservación de la diversidad biológica con el desarrollo económico (Young et al. 2005, Preisler et al. 2006). Las molestias de origen humano y sus efectos sobre el comportamiento en aves han sido un campo de estudio muy común (Steidl y Anthony 2000, Blumstein et al. 2005, Gill 2005, Zuberogoitia et al. 2008). La medida de conservación más habitualmente recomendada para evitar las molestias humanas ha sido el establecimiento de zonas de protección alrededor de áreas sensibles o críticas, en las que las actividades más lesivas son limitadas o prohibidas. Sin embargo, en el caso de actividades con una gran importancia económica y social (industriales, construcción de infraestructuras o explotación forestal), la implantación de medidas restrictivas genera y conlleva mayores dificultades (Donázar et al. 2002, Bautista et al. 2004, Speziale et al. 2008). Un ejemplo de este dilema es el caso del buitre negro y la extracción de corcho, una actividad tradicional de importancia socioeconómica desarrollada en varios países circunmediterráneos como España, Portugal, Marruecos, Francia, Italia y Argelia.

Los resultados del trabajo mostraron que el descorche afecta negativamente a la reproducción del buitre negro, tanto por el abandono del nido durante largos períodos de tiempo como por la reducción del éxito reproductivo. El abandono del nido por los adultos tuvo un impacto importante sobre los pollos, evidenciándose que los expuestos a la perturbación tuvieron una mortalidad tres veces más alta que en zonas donde no se desarrolló el descorche (aunque esta diferencia no resultó estadísticamente significativa). También se comprobó que las molestias por descorche redujeron un 20% el número de nidos exitosos. Del mismo modo, las actividades realizadas a menos de 500 m del nido provocaron una probabilidad de abandono del nido por los adultos de más del 25%, lo que sugiere que se deben extremar las precauciones cuando se trabaje a menos de 500 m para evitar molestias y, en consecuencia, el fracaso en la reproducción. Además de la distancia, el nivel de ruido también determinó la reacción de los buitres, una cuestión que previamente se ha demostrado como alteradora del comportamiento de las aves (Bowles 1995, Bautista et al. 2004, Arroyo y Razin 2006). La distancia a la que se produce con mayor probabilidad el abandono del nido se amplía si el nivel de ruido es bajo o se alcanza el silencio. Por último, aspectos como la realización de las actividades de extracción durante las horas más frescas del día, no prolongar el descorche alrededor de los mismos nidos durante más de un día consecutivo, o procurar desplazarse por la parte inferior de las laderas, debajo del nido, para que los buitres no perciban molestias de la parte superior, minimizan de manera contrastada el impacto del descorche.

La estrategia vital de una especie longeva como el buitre negro está caracterizada por una baja fecundidad y una elevada tasa de supervivencia (Cody 1972, Hiraldo 1983), de modo que la supervivencia adulta es el parámetro más condicionante sobre la dinámica poblacional (por ejemplo Meretsky et al. 2000, Oro et al. 2008). Así, podría argumentarse que los efectos de pérdida de productividad por molestias por descorche sobre la dinámica de la población pueden ser insignificantes, a lo que contribuye el hecho de que el descorche se produzca cada 9-10 años. La creación de áreas tampón en las que se prohíban o restrinjan actuaciones humanas puede constituir una buena herramienta de trabajo (González et al. 2006a), pero puede ser inviable o incluso negativo para la conservación del medio natural si las actividades humanas se convierten en no rentables económicamente y el abandono rural o el cambio de uso consecuentes pueden resultar más perjudiciales. En este sentido, el análisis costes-beneficios sugiere que los costes de eliminar o restringir el descorche son mayores que los beneficios que generaría para el buitre negro (en términos de aumento de la productividad). El hallazgo de protocolos que, manteniendo esta actividad, reduzcan los efectos negativos sobre los buitres debe considerarse prioritario. Así, es muy importante integrar la gestión de los espacios naturales con las actividades humanas que se desarrollan en ellos (Sergio et al. 2005), particularmente en el caso de especies que habitan áreas humanizadas. Además, los beneficios económicos generados por las actividades humanas pueden contribuir a preservar los ecosistemas en los que dichas actividades tienen lugar, como es el caso de los bosques maduros de alcornoque, explotados para la obtención del corcho. Del mismo modo, resulta recommendable examinar los costes y la eficacia de las actuaciones de conservación antes de proponer su aplicación (Sutherland et al. 2004, Pullin et al. 2004). Por último, este tipo de trabajos ayuda a reducir los posibles conflictos entre los distintos agentes involucrados en la gestión del medio natural mediante el análisis de los resultados y su discusión compartida.

Gestión de puntos de alimentación suplementaria siguiendo requisitos sanitarios y ecológicos

Las aves necrófagas proporcionan importantes servicios ecosistémicos y contribuyen a equilibrar las cadenas tróficas completando el ciclo de asimilación de la biomasa de animales muertos (DeVault et al. 2003, Şekercioğlu et al. 2004). Las especies necrófagas han desarrollado mecanismos evolutivos comportamentales y morfológicos que optimizan la explotación y aprovechamiento de cadáveres (König 1983, Donázar 1993, Hertel 1994). Otra cuestión importante relativa a las especies necrófagas y las carroñas, desde el punto de vista sanitario, es la interacción entre animales y seres humanos que puede provocar la expansión de determinadas enfermedades transmisibles (Caley y Hone 2004), y afectar negativamente a la salud animal y humana e incrementar los costes de gestión ganadera, de acciones de conservación y de medidas sanitarias (Daszak et al 2000, Horan y Wolf 2005, Gortázar et al. 2008, Ogada et al. 2012). Por ello, y para evitar riesgos sanitarios, es necesario que las técnicas de gestión de cadáveres de animales sean los más apropiadas y cuidadosas (Gortázar et al. 2007, Maichak et al. 2009).

La legislación europa relativa al manejo de subproductos animales no destinados a consumo humano (Reglamento CE 1774/2002) únicamente permitió, bajo determinadas prescripciones, la alimentación de aves necrófagas en puntos de alimentación vallados que evitaran la entrada de potenciales vectores de enfermedades transmisibles. La gestión y suministro de recursos alimenticios a través de muladares vallados se ha mostrado insuficiente, en términos de disponibilidad, para las poblaciones ibéricas de aves necrófagas (García de Francisco y Moreno-Opo 2009). De hecho, ha provocado alteraciones en los patrones de campeo, dispersión e incluso comportamiento (Deygout et al 2009, Margalida et al. 2010, Zuberogoitia et al. 2010), lo que ha conducido a cambios en la dinámica poblacional de las especies, la aparición de efectos negativos asociados a fuentes predecibles de alimento e impactos económicos y sociales (Carrete et al. 2006a, Robb et al. 2008; Margalida et al. 2011b). Por otro lado, es recomendable, en ciertas circunstancias, el empleo de sistemas y técnicas que eviten el acceso de necrófagos oportunistas potenciales vectores de enfermedades a las carroñas. En estos casos, y al objeto de proveer un rápido consumo de los subproductos por las aves necrófagas, sería necesario contar con modelos de cierres perimetrales que limiten el acceso únicamente a las especies objetivo.

El Capítulo 6 tuvo como finalidad contribuir a la compatibilización de los requisitos sanitarios y ecológicos en la alimentación suplementaria de aves necrófagas amenazas. Los objetivos específicos se dirigieron a testar modelos económicos, móviles y fácilmente manejables de cerramientos perimetrales que 1) evitaran la entrada a los aportes de carroña a mamíferos necrófagos facultativos, 2) replicaran de la manera más natural posible la aparición de carroña en el medio, y 3) aseguraran el consumo óptimo por parte de rapaces necrófagas. Los resultados mostraron que todas las especies de aves necrófagas accedieron a los aportes depositados en el interior de los cercados perimetrales móviles indistintamente del modelo que se tratara. Esto tiene su importancia para el desarrollo de protocolos de gestión de puntos de alimentación suplementaria; aunque la mayor parte de administraciones públicas abogan por la implantación de muladares convencionales (García de Francisco y Moreno-Opo 2009), los sistemas móviles también pueden satisfacer las necesidades de las aves necrófagas. Los muladares convencionales están formados por vallados perimetrales fijos que cumplen los requisitos sanitarios contra diseminación de enfermedades, aunque la previsibilidad en la oferta de carroña que ofrecen perjudica a los patrones naturales de búsqueda de alimento de las rapaces necrófagas.

En nuestro caso, los dos modelos móviles que evitaron más eficazmente la entrada de mamíferos necrófagos facultativos fueron las redes *alta* y *baja*, que mantuvieron durante alrededor de dos meses, en el mismo emplazamiento, su estanqueidad ante oportunistas como el zorro *Vulpes vulpes* en los aportes quincenales suministrados. Estos modelos son, además, recomendables por cumplir los requisitos iniciales del ensayo: fácil montaje y desmontaje por la no instalación de elementos fijos, y coste económico menor (acotar 0,5 ha con redes *alta* y *baja* cuesta aproximadamente 900 € respecto a los 6 000 €-9 000 € de los muladares convencionales).

Estos resultados experimentales son los primeros que muestran patrones de acceso a la carroña depositada en el interior de distintos tipos de cercados perimetrales por parte de distintas especies del gremio de necrófagos. En este sentido, dados los problemas relativos a la calidad y abundancia del alimento que afectan a las poblaciones de buitres del sur de Asia en los últimos años (Oaks et al. 2004, Gilbert et al. 2007) y que empiezan a detectarse y amenazar a las poblaciones africanas de buitres (Naidoo et al. 2009, Virani et al. 2011), la información ofrecida en este trabajo puede resultar de interés global para un adecuado manejo de los puntos de alimentación suplementaria (Markandya et al. 2008). Los modelos sugeridos pueden ser aplicados por los sectores sociales involucrados en la gestión de subproductos de ganado y de ungulados silvestres, de modo que ganaderos, cazadores y administraciones podrían aprovechar este tipo de modelos económicos y extrapolables a distintos escenarios geográficos. Esta colaboración, además, podría poner de relieve la utilidad de las aves necrófagas como aliados para el reciclado de la materia orgánica en el medio natural y disuadir del uso de métodos ilegales de control de depredadores que pudieran afectarlas negativamente (Koenig 2006, Hernández y Margalida 2008, 2009b, Margalida et al. 2011b).

Una perspectiva sobre la conservación del buitre negro en España

La evaluación del estado de conservación de especies amenazadas requiere distintas etapas de análisis, como son la identificación de las amenazas y sus impactos, el seguimiento de la población y el análisis de la eficacia de las medidas de gestión aplicadas (Soulé 1986, Groom et al. 2006). Estas tres cuestiones deben estar íntimamente relacionadas y sin una adecuada coordinación entre ellas es muy difícil obtener resultados exitosos (Pullin et al. 2004, Arlettaz et al. 2010). Cuando se pretende estudiar con perspectiva temporal cómo estas tres fases de análisis se han desarrollado, para conocer la historia de conservación de una especie, puede resultar de ayuda la recopilación de información bibliográfica y su consiguiente análisis a través de técnicas bibliométricas (Zhang et al. 2010, Liu et al. 2011). De esta manera, se pueden establecer correlaciones entre el esfuerzo de investigación y variables como la tendencia poblacional de una especie, las tasas de mortalidad no-natural o las medidas de conservación implantadas, permitiendo una discusión sobre el tipo de relaciones que pueden vincular dichas variables. Obviamente, existen numerosos aspectos que influyen en el devenir de una población de una especie amenazada y que deben tenerse en cuenta para concluir los términos causa-efecto de dichas relaciones. Entre estos aspectos, las actuaciones antropogénicas (envenemiento, molestias, alteración del hábitat, alimentación suplementaria, etc.) son las más determinantes (Oro et al. 2008, Ortega et al. 2009).

El trabajo expuesto en el Capítulo 7 constituye el primer análisis a largo plazo del esfuerzo de investigación y del estado de conservación del buitre negro, y probablemente uno de los pocos realizados para vertebrados amenazados en España. Las conclusiones obtenidas deben considerarse con cautela al existir sesgos en las relaciones causa-efecto que pudieran darse entre las variables estudiadas. Respecto a las amenazas, la consideración de la especie como protegida constituyó un hito importante en la década de 1970 que acabó con su persecución directa. Además, la protección de las áreas de reproducción, que cubre a más del 90% de las parejas existentes en España (Moreno-Opo y Guil 2007), aseguró la conservación de amplios territorios evitando acciones irreversibles alteradoras de los hábitats naturales. En relación a la disponibilidad de alimento, se ha evidenciado una correlación positiva entre el aumento de la población reproductora de buitre negro y las estadísticas de caza mayor, actividad que ha experimentado un auge muy importante en España en las últimas décadas (Garrido 2011). Por el contrario, los cambios en la regulación sobre subproductos animales acontecidos a principios de los años 2000, que condujeron a la recogida obligatoria de los cadáveres de ganado provocando una reducción de la disponibilidad de alimento (Donázar et al. 2009a, 2009b), no implicaron la disminución de la población reproductora de buitre negro. Del mismo modo, el declive general de las poblaciones de conejo de monte (Gortázar et al. 2007), una especie presa clave para la mayoría de depredadores amenazados de España (Delibes-Mateos et al. 2007), que provocó cambios en la dieta del buitre negro (Costillo et al. 2007a, b, Corbacho et al. 2007), no ha mostrado una asociación significativa con su tendencia poblacional.

El análisis bibliométrico mostró cómo los artículos publicados en revistas con impacto científico, así como los estudios sobre temas de conservación, amenazas y gestión y, en menor medida, los trabajos relacionados con la biología de la especie, aumentaron con una tendencia similar a la de la población reproductora de buitre negro. Este hecho podría poner de relieve que una mejor comprensión de los requerimientos ecológicos y de los efectos de las amenazas puede contribuir a un mayor éxito en las políticas de conservación (Pullin et al. 2004, Sutherland et al. 2004, Arlettaz et al. 2010) y, en consecuencia, conducir a una mejora de la situación poblacional de la especie. No obstante, resulta muy complejo analizar si existen vínculos causales entre el esfuerzo de investigación realizado y el estado de conservación o la tendencia poblacional (Pullin y Stewart 2006). En esta línea, para poder justificar causalidad entre ambas cuestiones, sería preciso un desfase entre la publicación del estudio, su aplicación y la verificación de efectos positivos. Así, podría discutirse que la publicación de trabajos científicos acontece tras la existencia de un adecuado conocimiento básico sobre la especie (Faaborg et al. 2010) e, incluso, como resultado de la propia recuperación de las poblaciones (lo que aumentaría las probabilidades de manejo de la especie por existir más individuos o por la mayor facilidad de obtención de permisos para investigar). Por ello, la información genérica y bien divulgada podría ser tan importante como la publicación de estudios científicos en revistas de impacto (Martín-López et al. 2009, Botrill et al. 2011).

El buitre negro en España puede constituir un ejemplo de coevolución entre el reconocimiento de los factores de amenaza y la aplicación de medidas de gestión, de manera que el conocimiento disponible ha permitido adaptar las políticas de conservación a las necesidades de la especie (McCarthy y Possingham 2007). Como resultado, por ejemplo, 1) se ha podido implantar un nuevo marco legal que permite acceder a carroñas de una manera sostenible y extensiva en el medio natural, a diferencia de la situación anterior donde únicamente se podía proporcionar alimento a las especies necrófagas en comederos vallados (European Commission 2011, Margalida et al. 2012a); 2) las actividades de aprovechamiento forestal en áreas de cría han sido evaluadas y se han implantado protocolos de actuación compatibles con la presencia de la especie sin necesidad de impedir dichas actividades (Moreno-Opo y Guil 2007, Margalida et al. 2011c); 3) se ha incrementado el conocimiento de los productos fitosanitarios más perjudiciales, promoviendo la prohibición de su uso y su sustitución por otros compuestos menos lesivos (Hernández y Margalida 2008, Council of Europe 2009b, 2009c); 4) los métodos de seguimiento de la especie se han evaluado, y se ha sugerido una homogeneización de los mismos para poder comparar los datos obtenidos en distintas regiones españolas (De la Puente et al. 2007); 5) se ha puesto de manifiesto la necesidad de proteger y gestionar adecuadamente las áreas de alimentación, como ámbito geográfico donde acontecen las principales amenazas para la especie, (Carrete y Donázar 2005, Moreno-Opo et al. 2010); 6) los principios de lucha contra el uso ilegal de cebos envenenados han sido actualizados, incorporando medidas prácticas informativas y preventivas para su seguimiento, persecución y castigo (Conover 2001, Dirección General para la Biodiversidad 2007); y 7) se ha indicado la utilidad y necesidad de contar con bases de datos sobre las distintas amenazas incidentes, especialmente el envenenamiento, como base documental para la priorización de medidas de conservación (Mateo 2010).

CONCLUSIONES

1. La conservación de las especies amenazadas requiere una aproximación científica inicial al reconocimiento de los requerimientos ecológicos de la especie, a la identificación de las amenazas y factores limitantes que la afectan y al análisis de la interacción entre amenazas y dinámica poblacional. Una vez logrado dicho conocimiento, la aplicación de medidas de actuación encaminadas a mejorar su estado de conservación tiene una mayor probabilidad de éxito.

2. Cuando el nivel de amenaza de una especie es muy elevado, a menudo es preciso iniciar la ejecución de acciones urgentes de conservación antes de haber conseguido el conocimiento necesario. Ello implicará que no se tengan las mayores garantías en la consecución de los objetivos planteados. En este caso, es preciso acompasar la investigación sobre biología, amenazas y patrones de selección de hábitat y alimentación de la especie, con la evaluación de la eficacia de las medidas de conservación aplicadas. Esta simbiosis permite una modulación de los planteamientos de manejo, inversión económica y líneas de actuación, en un marco de gestión adaptativa que resulta favorable y positivo.

3. El buitre negro es una de las especies para las que se ha destinado un mayor esfuerzo de investigación y un mayor número de actuaciones de gestión en la historia de la conservación de la naturaleza española. El incremento progresivo en la publicación de trabajos científicos principalmente relacionados con aspectos de la biología de la especie y de su manejo, ha ido acompañado de un aumento de la población reproductora en España. Ello podría evidenciar una relación entre conocimiento básico y diagnóstico de las mejores políticas de protección, aunque por el hecho de no poder probarse una relación causa-efecto entre ambas cuestiones conviene tomar con cautela cualquier conclusión al respecto.

4. La población de buitre negro ha aumentado significativamente en España en los últimos 40 años, habiéndose conseguido, secuencialmente en este período, la protección de los ejemplares, la gestión adecuada y sostenible de las áreas donde se reproduce y, en los últimos años, la adecuación legal conducente a mejorar la disponibilidad de alimento. No existen conclusiones claras sobre la dinámica de la mortalidad causada por la ingestión de cebos envenenados, principal factor de riesgo de la especie, que aparentemente muestra ciclos de intensidad sometidos a sesgos relacionados con la capacidad y esfuerzo de detección. Como retos futuros, resulta prioritario reducir la mortalidad no natural de ejemplares adultos y gestionar de manera adecuada los aprovechamientos económicos en las áreas donde los ejemplares campean y obtienen su alimento.

5. El buitre negro constituye un buen indicador de determinadas características de los hábitats donde se reproduce en España. Selecciona para criar masas boscosas maduras, con elevada cobertura vegetal, árboles altos, en laderas pronunciadas e inaccesibles para el ser humano. En general, a escala de colonia de cría, se registran mayores tasas de éxito reproductivo cuando los valores de las anteriores características

son más elevados. Por lo tanto, es recomendable continuar protegiendo los espacios con este tipo de características en las sierras y montes del centro, sur y oeste de España peninsular para garantizar la viabilidad de núcleos reproductores fuente.

6. Es preciso abordar de forma coordinada y homogénea la metodología de estudio de los patrones de selección de hábitat de una especie a distintas escalas geográficas. De esta manera, se producirán resultados comparables entre sí, que permitirán evaluar conjuntamente trabajos a escala global. Las propuestas de métodos de censo y seguimiento habrían de ser desarrolladas por las entidades, administraciones y/o expertos que trabajan de manera más directa con la especie, promoviendo coordinación a nivel nacional e internacional en la medida de lo posible.

7. Las actuaciones de aprovechamiento forestal desarrolladas en las colonias de cría de buitre negro influyen en su éxito reproductivo. Esta interacción se ha producido de manera secular en nuestros montes y, en la actualidad, existen bases de conocimiento y planes de actuación que la hacen compatible con la presencia del buitre negro. Resulta más eficiente proponer alternativas de actuación compatibles que prohibir los aprovechamientos y permitir que se visualice la especie objetivo como impedimento al desarrollo económico. Además, es preciso aunar esfuerzos para que los hábitats de reproducción del buitre negro se conserven y sean rentables para sus propietarios y gestores (caso de alcornocales y pinares maduros en explotación) y así reducir la posibilidad de que se altere el modelo de aprovechamiento existente o el abandono de las prácticas forestales tradicionales.

8. Los crotales identificativos que porta el ganado y que son ingeridos y posteriormente regurgitados por los buitres negros permiten averiguar localizaciones de zonas de alimentación de la especie (a escala de explotación agraria). Es un método novedoso y económico que permite determinar las áreas de campeo de los individuos pertenecientes a una colonia de cría, y que presenta diferencias metodológicas respecto a otros estudios destinados al mismo fin pero que emplean otras técnicas de seguimiento.

9. Las rapaces necrófagas reportan servicios ecosistémicos a través del consumo de carroñas, contribuyendo a un ahorro económico y energético y a una mayor salvaguarda sanitaria. En este sentido, el buitre negro depende de la ganadería extensiva y de la gestión cinegética de caza mayor, sectores a los que también favorece. Prueba de ello es que existe relación directa entre el crecimiento de la población de buitre negro y el de las estadísticas de caza de ungulados silvestres cinegéticos. Así, el mantenimiento de prácticas ganaderas tradicionales, sobre todo de ovino y caprino, y la gestión adecuada de las poblaciones de ungulados silvestres contribuyen a asegurar la disponibilidad de alimento para el buitre negro y éste, a su vez, actúa como coadyuvante para mantener la rentabilidad en las explotaciones y acotados.

10. La presencia del buitre negro en las carroñas depende del formato y disposición en que éstas se presentan. Si existen trozos dispersos de tamaño pequeño y mediano y en elevadas cantidades el número de ejemplares que acude a alimentarse es superior. De esta manera, y con el objetivo de evitar una proporción de necrófagas muy

desviada a favor del buitre leonado, que es la especie que acapara la mayor cantidad de recursos en las carroñas de aparición predecible, se recomienda depositar los restos de subproductos animales repartidos en una superficie lo más extensa posible y en trozos pequeños y medianos. Además, en la medida de lo posible, resulta conveniente incrementar los aportes durante la época de crianza del pollo (entre abril y agosto) que es cuando menor disponibilidad de alimento existe de forma natural en el medio mediterráneo.

11. En relación a la gestión de la alimentación de especies necrófagas, es preciso aplicar las medidas legales vigentes destinadas a procurar una aparición de carroña de la forma más dispersa e impredecible posible, asemejando las condiciones de aparición natural de cadáveres de animales silvestres y domésticos. En este sentido, se recomienda aplicar técnicas de gestión sostenibles y económicas que compatibilicen los requisitos de las aves necrófagas y la necesaria seguridad sanitaria en explotaciones ganaderas y acotados.

12. Es posible complementar la ejecución de trabajos de seguimiento de especies amenazadas, cuya finalidad principal es conocer aspectos demográficos, con el desarrollo de estudios conducentes a optimizar las políticas de conservación. En este sentido, resulta primordial planificar un adecuado diseño experimental y llevar a cabo un completo proceso formativo de técnicos de campo y guardería, para lograr la obtención de datos rigurosos que permitan alcanzar conclusiones solventes y comparables.

13. En definitiva, esta tesis doctoral expone información sobre distintos patrones de selección de recursos en ambientes de monte mediterráneo, en áreas de reproducción y de alimentación – en estas últimas considerando como unidad de estudio el propio alimento -, tomando como especie modelo al buitre negro. Tiene un enfoque eminentemente práctico e incluye en los análisis, en la medida de lo posible, la influencia humana a través de las actividades que desarrolla en el medio que comparte con la especie. Pretende, además, promover líneas compatibles de gestión del territorio, en un escenario de intervención de personas que tengan como objetivo, por un lado, favorecer el estado de conservación de la especie objetivo o, por otro, poder desarrollar actividades socioeconómicas en el medio rural. Las conclusiones y resultados obtenidos pueden ser aplicables a otros taxones que ocupan los mismos hábitats o presentan requerimientos ecológicos similares.

CONCLUSIONS

1. The conservation of endangered species requires an initial scientific approach aimed at the recognition of the ecological requirements of the species, the identification of limiting factors and the analysis of the interactions between threats and population dynamics. Once this knowledge is aquired, the application of measures to improve conservation status has a greater likelihood to succeed.

2. When the threat level to a species is very high, implementation of urgent conservation actions must often begin before obtaining required comprehensive knowledge. This implies that, at this stage, the best guarantees in achieving objectives are not ensured. In this case, it is necessary to consider research on biology, threats and patterns of habitat and feeding selection with the assessment of the effectiveness of conservation measures. This symbiosis allows a progressive modulation of the management approaches, the economic investment and priority policies, in a framework of adaptive management.

3. In the history of the Spanish nature conservation, the cinereous vulture is one of the species to which a great research effort and strong management actions have been devoted. The increase in the publication of scientific papers, mainly related to biological aspects, conservation and management, has been matched with the growth in its breeding population in Spain, showing a positive correlation between the increase in basic knowledge and the diagnosis of best practices to protect individuals and their habitats.

4. The cinereous vulture population has increased in Spain during the last 40 years, having sequentially achieved in this period the effective protection from intentional killing of the species, the sustainable management of breeding areas and in recent years legal enforcement leading to optimizing food availability. There are no reliable conclusions regarding the dynamics of mortality caused by poisoning, the main threat to the species, apparently showing cycles of intensity but subject to biases related to the ability of detection of the affected birds. The most important conservation challenges for the future are the reduction of non-natural mortality of adult birds and a proper management of the anthropogenic activities in foraging areas.

5. The cinereous vulture is a good indicator of certain characteristics of the habitats in which it breeds in Spain. For breeding, it selects mature forests with well-developed scrubland, large trees, on steep slopes inaccessible to humans. In general, at the breeding colony scale, greater breeding success is registered when the values of the above conditions are higher. Therefore, it is recommended that the protection and promotion of areas with such features in the mountains of central, southern and western Iberian Spain, continue to ensure the viability of core breeding areas.

6. A coordinated and consistent methodology to study habitat selection patterns of a species at different geographical scales must be addressed. Thus, it may be possible to obtain comparable results enabling the evaluatation of the results of

different studies on a global basis. The proposal of census and monitoring methods would be developed by institutions, governments and / or experts working more directly with the species, and promoting coordination at national and international levels as much as possible.

7. Forest harvesting activities developed within the breeding colonies of cinereous vultures influence breeding success adversely. This interaction has occurred secularly but, at present, there exists a foundation of knowledge to be applied to action plans that reconcile the presence of the cinereous vultures with logging activities in managed forests. The proposal of alternative methods for compatible use, such that people are not disadvantaged by the presence of an endangered species, can be more effecive than simply prohibiting harvesting practices, which could lead to the perception of target species as a deterrent to economic development. Furthermore, efforts should focus on the protection of breeding habitats and their maintenance in a way that remains profitable (such as the case of harvested cork oak and mature pine forests) such that habitat alteration is minimized andthe abandonment of traditional forestry practices is avoided.

8. Livestock ear tags allow the location of feeding points for vultures at the farm scale. They feed on the tags, along with the carrion, carrying them to the nests and perches and regurgitating them within their pellets. This is an novel technique to determine home ranges of individuals at a breeding colony, however cannot be used to make comparisons with other similar studies using different sampling methodology.

9. Avian scavengers provide ecosystem services through their carrion consumption by contributing to economic and energy savings and by increasing health guarantees. In this sense, the cinereous vulture both favors and depends on extensive livestock as well as on big game estates. This is evidenced in the concurrent growth of cinereous vulture breeding populations and wild ungulate hunting bags in recent years. Thus, maintaining traditional farming practices, particularly ovine enterprises, and the proper management of wild ungulate populations contributes to ensuring the availability of food for the cinereous vulture while this species, in turn, helps to balance the profitability of farmlands and hunting estates.

10. The presence of the cinereous vulture at carcasses depends on the form and manner of disposal in which carcasses occur. Thus, if there are small and medium scattered pieces in high quantities, the number of feeding cinereous vultures will be higher. In order to avoid a proportion of birds that is strongly biased in favor of the griffon vulture (the species monopolizing most feeding resources at carcasses) we recommend that remains of animal by-products be spread over a wide area and in small and medium pieces. Moreover, as much as possible, it is desirable to increase the input of food during the breeding season (between March and August) overlapping with the higher energy demand of the cinereous vultures and the lower food availability in the Mediterranean environments.

11. Regarding feeding management of scavenger raptors, it is necessary to implement the existing legal measures intended to ensure the occurrence of scattered

and unpredictable animal by-products as much as possible, simulating naturally-occurring conditions of carcasses. In this sense, it is recommended that sustainable and affordable management techniques are applied that reconcile ecological requirements of avian scavengers and the health security of livestock farms and hunting estates.

12. It is possible to complement the monitoring of endangered species, the main purpose of which is to inform demographic parameters, with a broader approach leading to the research of other ecological aspects, threats and best management practices. In this sense, it is essential to plan an appropriate experimental design and to implement a training process for field technicians and landholders, in order to achieve comparable and sound results.

13. In summary, this PhD Thesis presents information on patterns of the selection of resources by the cinereous vulture in Mediterranean environments, both in breeding and feeding areas. It has a practical focus and integrates within the analysis, as much as possible, human influence via activities carried out in the areas shared with the cinereous vultures. It also aims to promote compatible methods of land management, in a scenario of human intervention with targets that consider both the conservation of the cinereous vulture and, alternatively, the socio-economic development of rural areas.

BIBLIOGRAFÍA

Agudo, R., Rico, C., Hiraldo, F. and Donázar, J. A. (2011) Evidence of connectivity between continental and differentiated insular populations in a highly mobile species. *Diversity and Distributions* 17: 1-12.

Allen, A. P., Brown, J. H. and Gillooly, J. F. (2002) Global Biodiversity, Biochemical Kinetics and the Energetic-Equivalence Rule. *Science* 297: 1545–1548.

Arlettaz, R., Schaub, M., Fournier, J., Reichlin, T. S., Sierro, A., Watson, J. E. M. and Braunisch, V. (2010). From Publications to Public Actions: When Conservation Biologists Bridge the Gap between Research and Implementation. *Bioscience* 60: 835–842.

Armstrong, D. P. and Seddon, P. J. (2008) Directions in reintroduction biology. *Trends in Ecology and Evolution* 23: 20–25.

Arroyo, B.E. and Razin, M. (2006). Effects of human activities on bearded vulture behaviour and breeding success in the French Pyrenees. *Biological Conservation* 128: 276–284.

Aubad, J., Aragon, P. and Rodrıíguez, M. A. (2010) Human access and landscape structure effects on Andean forest bird richness. *Acta Oecologica* 36: 396–402.

Baillie, S. R. (1990) Integrated population monitoring of breeding birds in Britain and Ireland. *Ibis* 132: 151-166.

Barton, P. S., Cunningham, S. A., Lindenmayer, D. B. and Manning, A. D. (2012) The role of carrion in maintaining biodiversity and ecological processes in terrestrial ecosystems. *Oecologia.* DOI 10.1007/s00442-012-2460-3.

Bascompte, J. and Stouffer, D. B. (2009). The assembly and disassembly of ecological networks. *Philosophical Transactions of the Royal Society Biological Sciences* 364: 1781-1787.

Batbayar, N. (2012) The black vulture *Aegypius monachus* in southern and eastern Asia. In: Dobado, P. M. and Arenas, R. (cords.) *The Black Vulture: Status, Conservation and Studies*. Junta de Andalucía. Córdoba.

Bautista, L. M., García, J. T., Calmaestra, R. G., Palacín, C., Martín, C.A., Morales, M. B., Bonal, R. and Viñuela, J. (2004). Effect of weekend road traffic on the use of space by raptors. *Conservation Biology* 18: 1–7.

Benítez-López, A., Alkemade, R. and Verweij, P. A. (2010) The impact of roads and other infraestructures on mammal and bird populations: a meta-analysis. *Biological Conservation* 143: 1307–1316.

Bernis, F. (1966) El buitre negro *Aegypius monachus* en Iberia. *Ardeola* 12: 45-99.

Biostat (2006) *Comprehensive Meta-Analysis. A computer program for metaanalysis*. Englewood.

BirdLife International. (2004) *Birds in Europe. Population estimates, trends and conservation status*. BirdLife Conservation Series 12. BirdLife International. Cambridge.

BirdLife International. (2012). *Aegypius monachus; Neophron percnopterus; Gyps fulvus; Gypaetus barbatus*. In: IUCN. *IUCN Red List of Threatened Species*. Version 2012.1. www.iucnredlist.org. Downloaded on 17 September 2012.

Blumstein, D. T., Fernández-Juricic, E., Zollner, P.A. and Garity, S.C. (2005). Inter-specific variation of avian responses to human disturbance. *Journal of Applied Ecology* 42: 943–953.

Botrill, M. C., Walsh, J. C., Watson, J. E. M., Joseph, L. N. Ortega-Argueta, A. and Possingham, H. P. (2011). Does recovery planning improve the status of threatened species? *Biological Conservation* 144: 1595-1601.

Bowker, M. A., Miller, M. E., Belnap, J., Sisk, T. D. and Johnson, N. C. (2008) Prioritizing Conservation Effort through the Use of Biological Soil Crusts as Ecosystem Function Indicators in an Arid Region. *Conservation Biology* 22: 1523-1533.

Bowles, A.E. (1995) Responses of wildlife to noise. In Knight, R. and Gutzwiller, K. (Eds.) *Wildlife and Recreationists: Coexistence Through Management and Research.* Island Press. Washington. pp: 109-132.

Boyce, M. S. (1992) Population Viability Analysis. *Annual Review of Ecology and Systematics* 23: 481-506.

Boyd, J. and Banzhaf, S. (2007) What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* 63: 616-626.

Branton, M. and Richardson, J. S. (2011). Assessing the Value of the Umbrella-Species Concept for Conservation Planning with Meta-Analysis. *Conservation Biology* 25: 9-20.

Brooke, B. W., Sodhi, N. S. and Bradshaw, C. J. A. (2008) Synergies among extinction drivers under global change. *Trends in Ecology and Evolution* 23: 453-460.

Brooks, T., Da Fonseca, G. A. B. and Rodriguez, A. S. L. (2004) Species, Data and Conservation Planning. *Conservation Biology* 18: 1682-1688.

Brotons, L., Herrando, S. and Pla, M. (2007) Updating bird species distribution at large spatial scales: applications of habitat modelling to data from long-term monitoring programs. *Diversity and Distributions* 13: 276–288.

Caley, P. and Hone, H. (2004) Disease transmission between and within species and the implications in disease control. *Journal of Applied Ecology* 41: 94-104.

Camiña, A. (2007) Energía eólica y buitre negro. In: Moreno-Opo, R. and Guil, F. (Eds.) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

Carpenter, S. R. et al. (2009) Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciencies of the USA* 106: 1305-1312.

Carrete, M. and Donázar, J. A. (2005) Application of central-place foraging theory shows the importance of Mediterranean dehesas for the conservation of cinereous vulture *Aegypius monachus*. *Biological Conservation* 126: 582-590.

Carrete, M., Donázar, J. A. and Margalida, A. (2006a) Density-dependent productivity depression in Pyrenean Bearded Vultures: Implications for Conservation Plans. *Ecological Applications* 16: 1674-1682.

Carrete, M., Donázar, J. A., Margalida, A. and Bertrán, J. (2006b) Linking ecology, behaviour and conservation: does habitat saturation change the mating system of bearded vultures? *Biology Letters* 2: 624-627.

Carrete, M., Grande, J. M., Tella, J. L., Sánchez-Zapata, J. A., Donázar, J. A., Díaz-Delgado, R. and Romo, A. (2007) Habitat, human pressure, and social behavior: Partialling out factors affecting large-scale territory extinction in an endangered vulture. *Biological Conservation* 136: 143-154.

Carrete, M., Lambertucci, S. A. Speziale, K., Ceballos, O., Travaini, A., Delibes, M., Hiraldo, F. and Donázar, J. A. (2010) Winners and lossers in human-made habitats: interspecific competition outcomes in two Neotropical vultures. *Animal Conservation* 13: 390-398.

Carrete, M., Margalida, A., Oro, D. and Donázar, J. A. (2009b). Population consequences of vulture restaurants: the bearded vulture *Gypaetus barbatus* in the Spanish Pyrenees. In, Donázar, J. A., Margalida A. and Campión, D. (Eds.) Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology. *Munibe*, 29. Sociedad de Ciencias Aranzadi. Donostia.

Carrete, M., Sánchez-Zapata, J. A., Benítez, J. R., Donázar, J. A. and Lobón, M. (2009a) Large scale risk-assessment of wind-farms on population viability of a globally endangered long-lived raptor. *Biological Conservation* 142: 2954-2961.

Carrete, M., Sánchez-Zapata, J.A., Benítez, J.R., Lobón, M., Montoya, F. and Donázar, J.A. (2012) Mortality at wind-farms is positively correlated to large-scale distribution and aggregation in griffon vultures. *Biological Conservation* 145: 102-108.

Cody, M.L. 1971. Ecological aspects of reproduction. In: Farner D.S. and King J.R. (eds) *Avian Biology 1*. Academic Press. New York pp. 461-512.

Conover, M. R. (2001) *Resolving human-wildlife conflicts. The science of wildlife damage management*. Lewis Publishers. New York.

Corbacho, C., Costillo, E. and Morán, R. (2012) Home range and foraging area of Black vultures *Aegypius monachus* in south-western Spain: a preliminary analysis. In: Dobado, P. and Arenas, R. (coords.) *The Black Vulture: Status, Conservation and Studies*. Consejería de Medio Ambiente de la Junta de Andalucía. Córdoba, pp: 251-258.

Corbacho, C., Costillo, E. and Perales, A. B. (2007) La alimentación del buitre negro. In: Moreno-Opo, R. and Guil, F. (Eds.) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

Cortés-Avizanda, A. (2010) *The ecological and conservation effects of trophic resource predictability: carcasses and vertebrate communities.* PhD Thesis. Universidad Autónoma de Madrid.

Cortés-Avizanda, A., Carrete, M. and Donázar, J. A. (2010). Managing supplementary feeding for avian scavengers: guidelines for optimal design using ecological criteria. *Biological Conservation* 143: 1707-1715.

Costillo, E. (2005). *Biología y conservación de las poblaciones de buitre negro Aegypius monachus en Extremadura*. PhD Thesis. University of Extremadura. Badajoz.

Costillo, E., Corbacho, C., Morán, R. and Villegas, A. (2007a) The diet of the black vulture *Aegypius monachus* in response to environmental changes in Extremadura (1970-2000). *Ardeola* 54: 197-204.

Costillo, E., Corbacho, C., Morán, R. and Villegas, A. (2007b). Diet plasticity of Cinereous Vulture *Aegypius monachus* in different colonies in the Extremadura (SW Spain). *Ardea* 95: 201-211.

Costillo, E., Corbacho, C., Sánchez, J. M. and Villegas, A. (2007c). Áreas de campeo. In: Moreno-Opo, R. and Guil, F. (Eds.) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid. Pp. 132-141.

Council of Europe. (2009a) Regulation CE 1069/2009 of the Parliament and Council of Europe of 21st October. *Official Journal of the European Union* 300: 1-33.

Council of Europe. (2009b) Regulation CE 1107/2009 of the European Parliament and the Council of 21st October, concerning the placing of plant protection products on the market and repealing Council Directives 79/117/EEC and 91/414/EEC. *Official Journal of the European Union* 309: 1-50.

Council of Europe. (2009c) Directive 2009/128/EC of the European Parliament and of the Council of 21st October establishing a framework for Community action to achieve the sustainable use of pesticides. *Official Journal of the European Union* 309: 71-86.

Cramp, S. 1998. *The birds of the Western Palearctic on CD-Rom*. Oxford University Press. Oxford.

Cuevas, J. A. and De la Puente, J. (2005) *Hábitat potencial del buitre negro Aegypius monachus en la Sierra de Guadarrama*. Serie Documentos nº 45. CIA Fernando González Bernáldez. Madrid.

Dalley, K. L., Taylor, P .D. and Shutler, D. (2008) Nest-site characteristics and breeding success of three species of boreal songbirds in western Newfoundland, Canada. *Canadian Journal of Zoology* 86:1203-1211.

Daszak, P. and Cunningham, A. A. and Hyatt, A. D. (2000) Emerging infectious diseases of wildlife- threats to biodiversity and human health. *Science* 287: 443-449.

De Juana, E. and De Juana, F. (1984) Cabaña ganadera y distribución y abundancia de los buitres común *Gyps fulvus* y negro *Aegypius monachus* en España. In: Alamany, O. et al. (eds.) *Rapinyaires Mediterranis*. Centre de Recerca i Protecció de Rapinyaires. Barcelona, pp. 32-45.

De la Puente, J. (2006). Effect of monitoring frequency and date on estimates of abundance and productivity of colonial black vultures *Aegypius monachus* in Central Spain. In: Houston, D. C. and Piper, S. E. (Eds.) *Proceedings of the International Conference on Conservation and*

Management of Vulture Populations. Natural History Museum of Crete-WWF Greece. Thessaloniki.

De la Puente, J. (2007). Selección del hábitat de nidificación de buitre negro en pinares. In: Moreno-Opo, R. and Guil, F. (Coord.). *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid. pp. 53-57.

De la Puente, J. and Elorriaga, J. (2012). Primary moult and its application to ageing in the Black vulture *Aegypius monachus*. In: Dobado, P. M. and Arenas, R. (cords.) *The Black Vulture: Status, Conservation and Studies*. Junta de Andalucía. Córdoba.

De la Puente, J., Moreno-Opo, R. and Del Moral, J. C. (2007) *El buitre negro en España. Censo Nacional 2006*. SEO/BirdLife. Madrid.

De Lucas, M., Ferrer, M., Bechard, M. J. and Muñoz, A. R. (2012) Griffon vulture mortality at wind farms in southern Spain: Distribution of fatalities and active mitigation measures. *Biological Conservation* 147: 184-189.

Del Hoyo, J., Elliott, A. and Sargatal, J. (1994) *Handbook of the Birds of the World. Vol. 2. New World Vultures to Guineafowl.* Lynx Edicions. Barcelona.

Del Moral, J. C. (2009a) *El buitre leonado en España. Población reproductora en 2008 y método de censo*. SEO/BirdLife. Madrid.

Del Moral, J. C. (2009b) *El alimoche común en España. Población reproductora en 2008 y método de censo*. SEO/BirdLife. Madrid.

Del Moral, J. C. and De la Puente, J. (2010) Buitre negro – *Aegypius monachus*. In: Salvador, A. and Bautista, L. M. (Eds.). *Enciclopedia Virtual de los Vertebrados Españoles*. Museo Nacional de Ciencias Naturales. Madrid. http://www.vertebradosibericos.org/

Delibes-Mateos, M., Ferreras, P. and Villafuerte, R. (2009) European rabbit population trends and associated factors: a review of the situation in the Iberian Peninsula. *Mammal Review* 39: 124-140.

Delibes-Mateos, M., Redpath, S.M., Angulo, E., Ferreras, P. and Villafuerte, R. (2007). Rabbit as a keystone species in southern Europe. *Biological Conservation* 137: 149-156.

Demody, Bl J., Tanner, C. J. and Jackson, A. L. (2011) The Evolutionary Pathway to Obligate Scavenging in Gyps Vultures. *PLoS ONE* 6: e24635.

DeVault, T. L., Rhodes, O. E. and Shivik, J. A. (2003) Scavenging by vertebrates: behavioural, ecological and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. *Oikos* 102: 225-234.

Deygout, C., Gault, A., Duriez, O., Sarrazin, F. and Bessa-Gomes, C. (2010) Impact of food predictability on social facilitation by foraging scavengers. *Behavioral Ecology*. 21: 1131-1139.

Deygout, C., Gault, A., Sarrazin, F. and Bessa-Gomes, C. (2009) Modeling the impact of feeding stations on vulture scavenging service efficiency. *Ecological Modelling* 220: 1826-1835.

Díaz, S., Fargione, J., Chapin, S. and Tilman, D. (2006) Biodiversity Loss Threaten Human Well-Being. *Plos Biology* 4: e277. doi:10.1371/journal.pbio.0040277.

Dickman, A. J. (2010) Complexities of conflict: the importance of considering social factors for effectively resolving human–wildlife conflict. *Animal Conservation* 13: 458–466.

Dirección General para la Biodiversidad. (2005) *National Strategy against the illegal use of poisoned baits in the wild*. Ministry of Environment. Madrid.

Dobado, P. M. and Arenas, R. (cords.) *The Black Vulture: Status, Conservation and Studies*. Junta de Andalucía. Córdoba.

Donázar, J. A. (1992) Muladares y basureros en la biología y conservación de las aves en España. *Ardeola* 39: 29-40.

Donázar, J. A. (1993) *Los buitres ibéricos. Biología y Conservación*. Ed. J. M. Reyero. Madrid.

Donázar, J. A., Blanco, G., Hiraldo, F., Soto-Largo, E. and Oria, J. (2002) Effects of forestry and other land-use practices on the conservation of cinereous vultures. *Ecological Applications* 12: 1445-1456.

Donázar, J. A., Cortés-Avizanda, A. and Carrete, M. (2010) Dietary shifts in two cultures alter the demise of supplementary feeding stations: consequences of the EU sanitary legislation. *European Journal of Wildlife Research* 56: 613-621.

Donázar, J. A., Margalida, A. and Campión, D. (2009b) Vultures feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology. *Munibe* 29. Sociedad de Ciencias Aranzadi. San Sebastián.

Donázar, J. A., Margalida, A., Carrete, M. and Sánchez-Zapata, J. A. (2009a) Too sanitary for vultures. *Science* 326: 664.

Drew, C. A., Wiersma, Y. and Huettman, F. (2010) *Predictive species and habitat modelling in landscape ecology: concepts and applications*. Springer. London.

Dupont, H., Mihoub, J. B., Becu, N. and Sarrazin, F. (2011) Modelling interactions between scavenger behaviour and farming practices: Impacts on scavenger population and ecosystem service efficiency. *Ecological Modelling* 222: 982-992.

Engler, R., Guisan, A. and Rechsteiner, L. (2004) An improved approach for predicting the distribution of rare and endangered species from occurrence and pseudoabsence data. *Journal of Applied Ecology* 41: 263–274.

Estes, J. A. et al. (2011) Trophic downgrading of Planet Earth. *Science* 333: 301-306.

European Commission (1992) Council Directive 92/43/CE, of 21st May, on the conservation of natural habitats and of wild flora and Fauna. *Official Journal of the European Union* 206: 1–68.

European Commission (2011) Regulation CE 142/2011, of 25th February, implementing Regulation CE 1069/2009 of the European Parliament. *Official Journal of the European Union* 54: 1–254.

Faaborg, J. et al. (2010) Conserving migratory land birds in the New World: Do we know enough? *Ecological Applications* 20: 398-418.

Fargallo, J. A., Blanco, G. and Soto-Largo, E. (1998) Forest management effects on nesting habitat selected by european black vultures *Aegypius monachus* in central Spain. *Journal of Raptor Research* 32: 202-207.

Fernández-Juricic, E., Venier, M. P., Renison, D. and Blumstein, D. T. (2005). Sensitivity of wildlife to spatial patterns of recreationist behaviour: a critical assessment of minimum approaching distances and buffer areas for grassland birds. *Biological Conservation* 125: 225–235.

Fernández-Olalla, M., Martínez-Abraín, A., Canut, J., García-Ferré, D., Afonso, I. and González, L. M. (2012) Assessing different management scenarios to reverse the declining trend of a relict capercaillie population: A modelling approach within an adaptive management framework. *Biological Conservation* 148: 79-87.

Ferraro, PJ. and Pattanayak, S. K. (2006) Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. *PLoS Biology* 4: e105. doi:10.1371/journal.pbio.0040105.

Fielding, A. H. and Bell, J. F. (1997) A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation* 24: 38–49.

Finkelstein, M. E., Doak, D. F., George, D., Burnett, J., Brandt, J., Church, M., Grantham, J. and Smith, D. R. (2012) Lead poisoning and the deceptive recovery of the critically endangered California condor. *Proceedings of the National Academy of Sciences USA* 109: 11449-11454.

Fisher, B., Turner, R. K. and Morling, P. (2009) Defining and classifying ecosystem services for decision making. *Ecological Economics* 68: 643-653.

Forsman, D. (2003). *The raptors of Europe and the Middle East. A Handbook of Field Identification.* Christopher Helm. London.

Franklin, J. F. (1993) Preserving biodiversity: species, ecosystems or landscapes? *Ecological Applications* 3: 202-205.

Fundación Gypaetus (2005) *Plan de Acción de la Fundación Gypaetus contra los cebos envenenados*. Fundación Gypaetus. Jaén.

Gangoso, L., Álvarez-Lloret, P., Rodríguez-Navarro, A. B., Mateo, R., Hiraldo, F. and Donázar, J. A. (2009) Long-term effects of lead poisoning on bone mineralization in vultures exposed to ammunition sources. *Environmental Pollution* 157: 569-574.

García de Francisco, J. M. and Moreno-Opo, R. (2009) Livestock carcass management today: is there enough flexibility to deal with new conservation strategies? In: Donázar, J. A., Margalida, A. & Campión, D. (Eds.) Vultures, feeding stations and sanitary legislation: a conflict and its consequences from the perspective of conservation biology. *Munibe* 29. Sociedad de Ciencias Aranzadi. San Sebastián, pp. 502-509.

García-Ripollés, C. and López-López, P. (2011) Integrating effects of supplementary feeding, poisoning, pollutant ingestion and wind farms of two vulture species in Spain using a population viability analysis. *Journal of Ornithology* 152: 879-888.

Garrido, J. L. (2011). *Estimación de aprovechamientos cinegéticos en España por especies y comunidades autónomas*. Fedenca. Madrid.

Garzón, J. (1997) La importancia de la cultura agrarian para la conservación de la saves carroñeras. In: *Actas del II Congreso Internacional sobre Aves Carroñeras*. Solán de Cabras/Cañizares.

Gavashelishvili, A. and McGrady, M. J. (2006) Geographic information system-based modelling of vulture response to carcass appearance in the Caucasus. *Journal of Zoology* 269: 365-372.

Gavashelishvili, A., McGrady, M. J. and Javakhishvili, Z. (2006) Planning the conservation of the breeding population of cinereous vulture *Aegypius monachus* in the Republic of Georgia. *Oryx* 40: 1-8.

Gilbert, M., Watson, R.T., Ahmed, S., Asim, M. and Johnson, J.A. (2007) Vulture restaurants and their role in reducing diclofenac exposure in Asian vultures. *Bird Conservation International* 17: 63-77.

Gill, J.A. (2005) Approaches to measuring the effects of human disturbance on birds. *Ibis* 149: 9–14.

Gilman, E., Dunn, D., Read, A., Hyrenbach, K. D. and Warner, R. (2011) Designing criteria suites to identify discrete and networked sites of high value across manifestations of biodiversity. *Biodiversity and Conservation* 20: 3363-3383.

Glutz von Blotzheim, U., Bauer, K. and Bezzel, E. (1971) *Habdbuch der Vögel Mitteleuropas. Vol. 4: Falconiformes*. Frankfurt Am Main. Akademische Verlagsgesseslschaft.

Godfray, H. C. J. et al. (2010) Food Security: The Challenge of Feeding 9 Billion People. *Science* 327: 812- 818.

González, L. M., Arroyo, B. E., Margalida, A., Oria, J. and Sánchez, R. (2006a). Effect of human activities on the behaviour of breeding Spanish imperial eagles (*Aquila adalberti*): management implications for the conservation of a threatened species. *Animal Conservation* 9: 85–93.

González, L. M., Margalida, A., Sánchez, R. and Oria, J. (2006b) Supplementary feeding as an effective tool for improving breeding success on Spanish imperial eagle (*Aquila adalberti*). *Biological Conservation* 129: 477-486.

González, L. M. and Moreno-Opo, R. (2008) Impacto de la falta de alimento en las aves necrófagas amenazadas. *Ambienta* 73: 48-55.

González, L. M., Oria, J., Sánchez, R., Margalida, A., Aranda, A., Prada, L., Caldera, J. and Molina, J. I. (2008) Status and habitat changes in the endangered Spanish Imperial Eagle *Aquila adalberti* population during 1974–2004: implications for its recovery. *Bird Conservation International*18: 242–259.

Gortázar, C., Ferroglio, E., Höfle, U., Frölich, K. and Vicente, J. (2007) Diseases shared between wildlife and livestock: a European perspective. *European Journal of Wildlife Research* 53: 241-256.

Gortázar, C., Torres, M. J., Vicente, J., Acevedo, P., Reglero, M., De la Fuente, J., Negro, J. J. and Aznar, J. (2008) Bovine Tuberculosis in Doñana Biosphere Reserve: The Role of Wild Ungulates as Disease Reservoirs in the Last Iberian Lynx Strongholds. *PLoS ONE* 3: e2776.

Grande, J. M., Serrano, D., Tavecchia, G., Carrete, M., Ceballos, O., Díaz-Delgado, R., Tella, J. L. and Donázar, J. A. (2009). Survival in a long-lived territorial migrant: effects of life-history traits and ecological conditions in wintering and breeding areas. *Oikos* 118: 580-590.

Green, R. E., Newton, I., Shultz, S., Cunningham, A. A., Gilbert, M., Pain, D. J. and Prakash, V. (2004) Diclofenac poisoning as a cause of vulture population declines across the Indian subcontinent. *Journal of Applied Ecology* 41: 793-800.

Groom, M. J., Meffe, G. K. and Carroll, C. R. (2006). *Principles in Conservation Biology. Third Edition*. Sinauer Associates, Inc. Sunderland.

Grupo de trabajo del Quebrantahuesos (2012) *Informe nacional 2011 sobre el quebrantahuesos*. Ministerio de Agricultura, Alimentación y Medio Ambiente-Comunidades Autónomas. Informe inédito.

Guil, F. et al. (2011) Minimising Mortality in Endangered Raptors Due to Power Lines: The Importance of Spatial Aggregation to Optimize the Application of Mitigation Measures. *PLoS ONE* 6: e28212. doi:10.1371/journal.pone.0028212.

Guisan, A. and Thuiller, W. (2005) Predicting species distribution: offering more than simple habitat models. *Ecology Letters* 8: 993–1009.

Harris, G. M., Jenkins, C. N. and Pimm, S. L. (2005) Refining biodiversity conservation priorities. *Conservation Biology* 19: 1957-1968.

Hassan, R., Scholes, R. y N. Ash. 2005. *Ecosystems and human well-being: current state and trends*. Findings of the Condition and Trends Working Group. Island Press. Washington D.C.

Hazzah, L., Mulder, L. B. and Frank, L. (2009) Lions and Warriors: Social factors underlying declining African lion populations and the effect of incentive-based management in Kenya. *Biological Conservation* 142: 2428-2437.

Hernández, M. and Margalida, A. (2008). Pesticide abuse in Europe: effects on the Cinereous vulture *Aegypius monachus* population in Spain. *Ecotoxicology* 17: 264–272.

Hernández, M. and Margalida, A. (2009a) Assessing the risk of lead exposure for the conservation of the endangered Pyrenean bearded vulture (*Gypaetus barbatus*) population. *Environmental Research* 109: 837-842.

Hernández, M. and Margalida, A. (2009b). Poison-related mortality effects in the endangered Egyptian Vulture (*Neophron percnopterus*) population in Spain: conservation measures. *European Journal of Wildlife Research* 55: 415-423.

Hertel, F. (1994) Diversity in body size and feeding morphology within past and present vulture assemblages. *Ecology* 75: 1074-1084.

Heydon, M. J., Wilson, C. J. and Tew, T. (2010) Wildlife conflict resolution: a review of problems, solutions and regulation in England. *Wildlife Research* 37: 731-748.

Hijmans, R.J. and Graham, G. H. (2006) The ability of climate envelope models to predict the effect of climate change on species distributions. *Global Change Biology* 12: 2272–2281.

Hiraldo, F. (1974) Colonias de cría y censo de los buitres negros *Aegypius monachus* en España. *Naturalia Hispanica* 2. ICONA. Madrid.

Hiraldo, F. (1977) *El buitre negro Aegypius monachus en la península Ibérica. Población, biología general, uso de recursos e interacciones con otras aves*. PhD Thesis. University of Sevilla.

Hiraldo, F. (1983). Breeding biology of the cinereous vulture. In: Wilbur, S. R. and Jackson, J. A. (Eds.) *Vulture Biology and Management*. University of California Press. Berkeley.

Hiraldo, F. and Donázar, J. A. (1990) Foraging time in the cinereous vulture *Aegypius monachus*: seasonal and local variations and influence of weather. *Bird Study* 37: 128-132.

Hirzel, A. and Guisan, A. (2002) Which is the optimal sampling strategy for habitat suitability modelling. *Ecological Modelling* 157: 331–341.

Hoeksema, J. D. and Forde, S. E. (2008) A Meta-Analysis of Factors Affecting Local Adaptation between Interacting Species. *The American Naturalist* 171: 275–290.

Hoffman, M. et al. (2010) The Impact of Conservation on the Status of the World's Vertebrates. *Science* 330: 1503-1509.

Horan, R. D. D. and Wolf, C. A. A. (2005) The Economics of Managing Infectious Wildlife Disease. *American Journal of Agricultural Economics* 87: 537-551.

IUCN. (2001) *IUCN Red List Categories and Criteria: Version 3.1*. IUCN Species Survival Commission. Gland- Cambridge.

IUCN. (2003) *Guidelines for Application of IUCN Red List Criteria at Regional Levels: Version 3.0*. IUCN Species Survival Commission. Gland-Cambridge.

Jackson, A. L., Ruxton, G. D. and Houston, D. C. (2008) The effect of social facilitation on foraging success in vultures: a modelling study. *Biology Letters* 4: 311-313.

Jiménez, J. (2002) Elaboración de un modelo de hábitat de reproducción de buitre negro *Aegypius monachus* en los Montes de Toledo. In: Torralvo, C. (ed.) *Anuario Ornitológico de Ciudad Real 1995–2001*. SEO-Ciudad Real. Ciudad Real.

Jiménez, J. and González, L. M. (2012). Patrones de movimiento y uso del espacio en la dispersión juvenil del buitre negro (*Aegypius monachus*). *Ecología* 24: 73-93.

Jiménez-Valverde, A., Lobo, J. M. and Hortal, J. (2008) Not as good as they seem: the importance of concepts in species distribution modelling. *Diversity and Distributions* 14: 885–890.

Jones, G. G. (2004) Conservation management of endangered birds. In Sutherland, W.J., Newton, I. & Green, R.E (eds.) *Bird Ecology and Conservation: a Handbook of Techniques*. Oxford University Press. Oxford. pp. 269–301.

Katzner, T., Millner-Guland, E. J. and Bragin, E. (2007) Using modelling to improve monitoring of structured populations: Are we collecting the right data? *Conservation Biology* 21: 241–252.

Klein, C., Wilson, K., Watts, M., Stein, J., Berry, S., Carwardine, J., Smith, M. S., Mackey, B. and Possingham, H. (2009) Incorporating ecological and evolutionary processes into continental-scale conservation planning. *Ecological Applications* 19: 206-207.

Koenig, R. (2006) Vulture research soars as the scavengers' numbers decline. *Science* 312: 1591–1592.

König, C. (1983) Interspecific and intraspecific competition for food among Old World vultures. In: Wilbur, S.R. and Jackson, J.A. (Eds.) *Vulture biology and management*. University ofCalifornia Press. Berkeley. pp. 153-171.

Kramer-Schadt, S., Revilla, E. and Wiegand, T. (2005) Lynx reintroductions in fragmented landscapes of Germany: projects with a future or misunderstood wildlife conservation? *Biological Conservation* 125: 169-182.

Kremen, C. (2005) Managing ecosystem services: what do we need to know about their ecology? *Ecology Letters* 8: 468-479.

Kostrzewa, R. and Kostrzewa, A. (1991) Winter weather, spring and summer density, and subsequent breeding success or Eurasian kestrels, common buzzards, and northern goshawk. *The Auk* 108: 342-347.

Lambeck, R. J. (1997) Focal Species: a Multi-Species Umbrella for Nature Conservation. *Conservation Biology* 11: 849-856.

Larsen, F.W., Bladt, J. and Rahbek, C. (2007) Improving the performance of indicator groups for the identification of important areas for species conservation. *Conservation Biology* 21: 731–740.

Lascroel, A., Dugger, K. M., Ballard, G. and Ainly, D. G. (2009) Effects of individual quality, reproductive success and environmental variability on survival of a long-lived seabird. *Journal of Animal Ecology* 78: 798-806.

Likens, G. E. and Lindenmayer, D. B. (2012) Integrating approaches leads to more effective conservation of biodiversity. *Biodiversity and Conservation* 21: 3323–3341.

Lind, J. and Creswell, W. (2005) Determining the fitness consequences of antipredation behavior. *Behavioural Ecology* 16: 945-956.

Lindenmayer, D. B. et al. (2008) A checklist for ecological management of landscapes for conservation. *Ecology Letters* 11: 78-91.

Lindenmayer, D. B. and Likens, G. E. (2011) Direct Measurement versus Surrogate Indicator Species for Evaluating Environmental Change and Biodiversity Loss. *Ecosystems* 14: 47-59.

Lindenmayer, D. B., Margules, C. R. and Botkin, D.B. (2000) Indicators of biodiversity for ecologically sustainable forest management. *Conservation Biology* 14: 941–950.

Liu, X., Zhang, L. and Hong, S. (2011). Global biodiversity research during 1900–2009: a bibliometric análisis. *Biodiversity and Conservation* 20: 807–826.

López-Bao, J. V., Rodríguez, A. and Palomares, F. (2008) Behavioural response of a trophic specialist, the Iberian lynx, to supplementary food: patterns of food use and implications for conservation. *Biological Conservation* 141: 1857-1867.

LPO (Ligue de protection des oiseaux). 2010. *Le vautour moin ; Suivi et conservation*. http://rapaces.lpo.fr/vautour-moine.

Luck, G.W. et al. (2004) Alleviating spatial conflict between people and biodiversity. *Proceedings of the National Academy of Sciences of the USA* 101: 182–186.

Luque, E., Dobado, P. and Arenas, R. (2010). Reproducciones atípicas del buitre negro en Andalucía. *Quercus* 291: 48-49.

Mace, G. M., Collar, N. J., Gaston, K, J., Hilton-Taylor, G., Akçakaya, H. R., Leader-Williams, N., Miller-Gulnard, E. J. and Stuart, S. N. (2008) Quantification of Extinction Risk: IUCN's System for Classifying Threatened Species. *Conservation Biology* 22: 1424-1442.

Mace, G. M., Gittleman, J. L. and Purvis, A. (2003) Preserving the Tree of Life. *Science* 300: 1707-1709.

Madroño, A., González, C. and Atienza, J. C. (Eds) (2004) *Libro Rojo de las Aves de España*. SEO/BirdLife-Ministerio de Medio Ambiente. Madrid.

Maes, J., Paracchini, M. L., Zulian, G., Dunbar, M. B. and Alkemade, R. (2012) Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. *Biological Conservation* 155: 1-12.

Magana, M., Alonso, J. C., Martín, C. A., Bautista, L. M. and Martín, B. (2010) Nest-site selection by Great Bustards *Otis tarda* suggests a trade-off between concealment and visibility. *Ibis* 152: 77–89.

Magurran, A. E. et al. (2010) Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends in Ecology and Evolution* 25: 574-582.

Maichak, E. J., Scurlock, B. M., Rogerson, J. D., Meadows, L. L., Barbknecht, A. E., Edwards, W. H. and Cross, P. C. (2009) Effects of management, behaviour, and scavenging on risk of brucellosis transmission in elk of western Wyoming. *Journal of Wildlife Diseases* 45: 398-410.

Margalida, A. (2010a) *Conservation biology of the last and largest natural population of the European bearded vulture (Gypaetus barbatus) (Linnaeus, 1758).* PhD Thesis. University of Bern. Bern.

Margalida, A. (2010b) Supplementary feeding during the chick-rearing period is ineffective in increasing the breeding success in the bearded vulture (*Gypaetus barbatus*). *European Journal of Wildlife Research* 56: 673–678.

Margalida, A. (2012). Baits, budget cuts: A deadly mix. *Science* 338: 192.

Margalida, A., Benítez, J. R., Sánchez-Zapata, J. A., Ávila, E., Arenas, R. and Donázar, J. A. (2012b) Long-term relationship between diet breadth and breeding success in a declining population of Egyptian Vultures *Neophron percnopterus*. *Ibis* 154: 184-188.

Margalida, A. and Bertran, J. (2000) Nest-building behaviour of the Bearded Vulture *Gypaetus barbatus*. *Ardea* 88: 259-264.

Margalida, A. Bertran, J. and Heredia, R. (2009) Diet and food preferences of the endangered Bearded vulture *Gypaetus barbatus*: a basis for their conservation. *Ibis* 151: 235-243.

Margalida, A., Campión, D. and Donázar, J. A. (2011b) European vultures' altered behaviour. *Nature* 480: 457.

Margalida, A., Carrete, M., Sánchez-Zapata, J. A. and Donázar, J. A. (2012a) Good news for European vultures. *Science* 335: 284.

Margalida, A. and Colomer, M. A. (2012) Modelling the effects of sanitary policies on European vulture conservation. *Scientific Reports* 2: 753.

Margalida, A., Colomer, M.A. and Sanuy, D. (2011a) Can wild ungulate carcasses provide enough biomass to maintain avian scavenger populations? An empirical assessment using a bio-inspired computational model. *PLoS ONE* 6: e20248.

Margalida, A., Donázar, J.A., Bustamante, J., Hernández, F. and Romero-Pujante, M. (2008a) Application of a predictive model to detect long-term changes in nest-site selection in the Bearded Vultures: conservation in relation to territory shrinkage. *Ibis* 150: 242-249.

Margalida, A., Donázar, J. A, Carrete, M. and Sánchez-Zapata, J. A (2010) Sanitary versus environmental policies: fitting together two pieces of the puzzle of European vulture conservation. *Journal of Applied Ecology* 47: 931–935.

Margalida, A., González, L. M., Sánchez, R., Oria, J., Prada, L., Caldera, J., Aranda, A. and Molina, J. I. (2007) A long-term scale study of the breeding biology of Spanish imperial eagle *Aquila adalberti*. *Journal of Ornithology* 148: 309-322.

Margalida, A., Heredia, R., Razin, M. and Hernández, M. (2008b). Sources of variation in mortality of the BeardedVulture *Gypaetus barbatus* in Europe. *Bird Conservation International* 18: 1-10.

Margalida, A., Moreno-Opo, R., Arroyo, B. E. and Arredondo, A. (2011c). Reconciling the conservation of endangered species with economically important anthropogenic activities: interactions between cork exploitation and the cinereous vulture *Aegypius monachus* in Spain. *Animal Conservation* 14: 167-174.

Margules, C. R. and Presey, R. L. (2000) Systematic conservation planning. Nature 405: 243-253.

Markandya, A., Taylor, T., Longo, A., Murty, M. N., Murty, S. and Dhavala, K. (2008) Counting the cost of vulture decline— An appraisal of the human health and other benefits of vultures in India. *Ecological Economics* 67:194–204.

Martín-López, B., Montes, C., Ramírez, L. and Benayas, J. (2009) What drives policy decision-making related to species conservation? *Biological Conservation* 149: 1370-1380.

Martínez-Abraín, A., Oro, D., Jiménez, J., Stewart, G. and Pullin, A. (2010) A systematic review of the effects of recreational activities on nesting birds of prey. *Basic and Applied Ecology* 11:312-319.

Martínez-Abraín, A., Tavecchia, G., Regan, J. M., Jiménez, J., Surroca, M. and Oro, D. (2012) Effects of wind farms and food scarcity on a large scavenging bird species following an epidemic of bovine spongiform encephalopathy. *Journal of Applied Ecology* 49: 109-117.

Mateo, R. (2010) Toxicology and wildlife conservation in Europe: the inadequacy of current EU regulations. *The Veterinary Journal* 183: 241-242.

Maxwell, D. and Jennings, S. (2005) Power of monitoring programmes to detect decline and recovery of rare and vulnerable fish. *Journal of Applied Ecology* 42: 25-37.

McCarthy, M. A. and Possingham, H. P. (2007) Active Adaptive Management for Conservation. *Conservation Biology* 21: 956–963.

Meffe, G.K. and Carroll, C.R. (1994) *Principles of Conservation Biology*. Sinauer Associates, Inc. Massachusetts.

Meffe, G. K., Nielsen, L. A., Knight, R. L. and Schenborn, D. A. (2002) *Ecosystem management*. Isalnd Press. Washington D. C.

Meretsky, V., Snyder, N., Beissinger, S. and Clendenen, D.J. (2000) Demography of the California condor: implications for reestablishment. *Conservation Biology* 14: 957–967.

Ministerio de Medio Ambiente, y Medio Rural y Marino (2010) *Situación de las poblaciones de rapaces necrófagas en relación a la disponiblidad de alimento.*Informe inédito. Madrid.

Ministerio de Medio Ambiente, y Medio Rural y Marino (2011) Real Decreto 139/2011, de 4 de febrero, para el desarrollo del Listado de Especies Silvestres en Régimen de Protección Especial y del Catálogo Español de Especies Amenazadas. *Boletín Oficial del Estado* 46: 20912-29951.

Minteer, B. A. and Collins, J. P. (2005) Why we need an "ecological ethics". *Frontiers in Ecology and Environment* 3: 332-337.

Morán, R., Sánchez, J. M., Costillo, E. and Villegas, A. (2007). El buitre negro en masas de monte mediterráneo. In: Moreno-Opo, R. and Guil, F. (Eds.) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid. pp. 179-196.

Morán-López, R., Sánchez, J. M., Costillo, E., Corbacho, C. and Villegas A. (2006) Spatial variation in anthropic and natural factors regulating the breeding success of the cinereous vulture *Aegypius monachus* in SW Iberian Peninsula. *Biological Conservation* 130: 169-182.

Morán-López, R., Sánchez, J. M., Costillo, E. and Villegas, A. (2005) Nest-site selection of endangered cinereous vulture *Aegypius monachus* populations affected by anthropogenic disturbance: present and future conservation implications. *Animal Conservation* 9: 29-37.

Moreno-Opo, R., Arredondo, A. and Guil, F. (2010) Foraging range and diet of cinereous vulture *Aegypius monachus* using livestock resources in central Spain. *Ardeola* 57: 111-119.

Moreno-Opo, R., Fernández-Olalla, M., Margalida, A., Arredondo, A. and Guil, F. (2012b). Effect of methodological and ecological approaches on heterogeneity of nest-site selection of a long-lived vulture. *PLoS ONE* 7: e33469.

Moreno-Opo, R., and Guil, F. (2007) *Manual de gestión del hábitat y de las poblaciones de buitre negro en España*. Ministerio de Medio Ambiente. Madrid.

Moreno-Opo, R., Guil, F., El Khadir, N., Oria, J., Soria, C. and Guzmán, J. (2012a) Conservation of Black vulture *Aegypius monachus* in private-owned estates. In: Dobado, P. M. and Arenas, R. (cords.) *The Black Vulture: Status, Conservation and Studies*. Junta de Andalucía. Córdoba.

Moreno-Opo, R. and Margalida, A. In rev. Conservation of the cinereous vulture *Aegypius monachus* in Spain (1966-2011): threats, research and adaptive management.

Moreno-Opo, R., Margalida, A., García, F., Arredondo, A., Rodríguez, C. and González, L. M. (2012c) Linking sanitary and ecological requirements in the management of avian scavengers: effectiveness of fencing against mammals in supplementary feeding sites. *Biodiversity and Conservation* 21:1673–1685

Moritz, C. 1994. Defining evolutionary significant units for conservation. *Trends in Ecology and Evolution* 9:373-375.

Moritz, C. 2002. Strategies to protect biological diversity and the evolutionary processes that sustain it. *Systematic Biology* 51: 238-254.

Moss, R., Watson, A. and Ollason, J. (1982) *Animal Population Dynamics*. Chapman and Hall. London.

Naidoo, V., Wolter, K., Cuthbert, R. and Duncan, N. (2009) Veterinary diclofenac threatens Africa's endangered vulture species. *Regulatory Toxicology and Pharmacology* 53: 205-208.

New, L. F., Buckland, S. T., Redpath, S. and Matthiopoulos, J. (2011) Han harrier management: insights from demographic models fitted to population data. *Journal of Applied Ecology* 48: 1187-1194.

Newton, I. (1979) *Population ecology of raptors*. T & AD Poyser. London.

Noguera, J. C., Pérez, I. and Mínguez, E. (2010) Impacto f terrestrial wind farms on diurnal raptors: developing a spatial vulnerability index and potencial vulnerability maps. *Ardeola* 57: 41-53.

Novacek, M. J. and Cleland, E. E. (2001) The current biodiversity extinction event: scenarios for mitigation and recovery. *Proceedings of the National Academy of Sciences of the USA* 98: 5466-5470.

Oaks, J. L. et al. (2004) Diclofenac residues as the cause of vulture population decline in Pakistan. *Nature* 427: 630-633.

Ogada, D. L., Torchin, M. E., Kinnaird, M. F. and Ezenwa, V. O. (2012) Effects of Vulture Declines on Facultative Scavengers and Potential Implications for Mammalian Disease Transmission. *Conservation Biology* DOI: 10.1111/j.1523-1739.2012.01827.x.

O´Grady, et al (2004) Correlations among extinction risks assessed by different systems of threatened species categorization. *Conservation Biology* 18: 1624-1635.

Olea, P. P. and Mateo-Tomás, P. (2009) The role of traditional farming practices in ecosystem conservation: The case of transhumance and vultures. *Biological Conservation* 142: 1844-1853.

Olea, P., P. and Mateo-Tomás, P. (2011) Spatially explicit estimation of occupancy, detection probability and survey effort needed to inform conservation planning. *Diversity and Distributions* 17: 714–724.

Oro, D., Margalida, A., Carrete, M., Heredia, R. and Donázar, J. A. (2008) Testing the Goodness of Supplementary Feeding to Enhance Population Viability in an Endangered Vulture. *PLoS ONE* 3: e4084.

Ortega, E., Mañosa, S., Margalida, A., Sánchez, R., Oria J. and González, L.M. (2009) A demographic description of the recovery of the vulnerable Spanish Imperial Eagle *Aquila adalberti*. *Oryx* 43: 113-121.

Paillet, Y. et al. (2009) Biodiversity differences between managed and unmanaged forests: Meta-analysis of species richness in Europe. *Conservation Biology* 24: 101–112.

Parra, J. and Tellería, J. L. (2004) The increase in the Spanish population of Griffon Vultur*e Gyps fulvus* during 1989–1999: effects of food and nest site availability. *Bird Conservation International* 14: 33-41.

Pérez, M. C. (2007) Diversity in the cork oak forests of the northern Straits of Gibraltar Region (Southern Spain). *Plant Ecology* 189: 87–100.

Poizaridis, K., Goutner, V., Skartsi, T. & Stamou, G. (2004) Modelling nesting habitat as a conservation tool for the Eurasian black vulture *Aegypius monachus* in Dadia Nature Reserve, northeastern Greece. *Biological Conservation* 118: 235-248.

Preisler, H.K., Ager, A. and Wisdom, M.J. (2006) Statistical methods for analysing responses of wildlife to human disturbance. *Journal of Applied Ecology* 43: 164–172.

Primack, R. B. (2012) *A primer of conservation biology*. Fifth Edition. Sinauer Associates, Inc. Massachusetts.

Pullin, A.S. (2002) *Conservation biology*. Cambridge University Press. Cambridge.

Pullin, A. S. and Knight, T. M. (2005) Assessing Conservation Management's Evidence Base: a Survey of Management-Plan Compilers in the United Kingdom and Australia. *Conservation Biology* 19: 1989-1996.

Pullin, A. S., Knight, T. M., Stone, D. A. and Charman, K. (2004). Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation* 119: 245-252.

Pullin, A. S. and Stewart, G. B. (2006) Guidelines for systematic review in conservation and environmental management. *Conservation Biology* 20: 1647–1656.

Redford, K. H. et al. (2011) What Does It Mean to Successfully Conserve a (Vertebrate) Species? *BioScience*, 61: 39-48.

Richardson, C. T. and Miller, C. K. (1997) Recommendations for protecting raptors from human disturbance: a review. *Wildlife Society Bulletin* 25: 634–638.

Robb, G. N., McDonald, R. A., Chamberlain, D.E. and Bearhop, S. (2008) Food for thought: supplementary feeding as a driver of ecological change in avian populations. *Frontiers in Ecology and Environment* 6: 476-484.

Robinson, S., Chiaradia, A. and Hindell, M.A. (2005) The effect of body condition on the timing and success of breeding little penguins *Eudryptula minor*. *Ibis* 147:483-489.

Roberge, J.M. and Angelstam, P. (2004) Usefulness of the umbrella species concept as a conservation tool. *Conservation Biology* 18: 76–85.

Rockström, J. et al. (2009) A safe operating space for humanity. *Nature* 461: 472-475.

Ruxton, G. D. and Houston, D. C. (2004) Obligate vertebrate scavengers must be large soaring fliers. *Journal of Theorethical Biology* 228: 431-436.

Sala, O. E. et al. (2000) Global Biodiversity Scenarios for the Year 2100. *Science* 287: 1770-1774.

Salafsky, N., Margoluis, R. and Redford, K. (2001). *Adaptive Management: A Tool for Conservation Practitioners.* Biodiversity Support Program. Washington, D.C.

Sánchez, J. J. (2004) Buitre negro *Aegypius monachus*. In: Madroño, A., González, C. and Atienza, J. C. (Eds.) *Libro rojo de las aves de España*. Ministerio de Medio Ambiente-SEO/BirdLife. Madrid.

Sanz-Aguilar, A., Tavecchia, G., Pradel, R., Mínguez, E. and Oro, D. (2008) The cost of reproduction and experience-dependent vital rates in a small petrel. *Ecology* 89: 3195-3203.

Schadt, S. et al. (2002) Assessing the suitability of central European landscapes for the reintroduction of Eurasian lynx. *Journal of Applied Ecology* 39: 189-203.

Schröter, T. et al. (2005) Ecosystem Service Supply and Vulnerability to Global Change in Europe. *Science* 310: 1335-1337.

Scott, J. M., Heglund, P. J. and Morrison, M. L. (2002) *Predicting species occurrences: issues of accuracy and scales*. Island Press. Washington, DC.

Seddon, P.H. and Leech, T. (2008) Conservation short cut, or long and winding road? A critique of umbrella species criteria. *Oryx* 42: 240–245.

Segurado, P. and Araujo, M. B. (2004) An evaluation of methods for modelling species distributions. *Journal of Biogeography* 31: 1555–1568.

Şekercioğlu, Ç. H. (2006) Increasing awareness of avian ecological function. *Trends in Ecology and Evolution* 21: 464-471.

Şekercioğlu, Ç. H., Daily, G. C. and Ehrlich, P. R. (2004) Ecosystem consequences of bird declines. *Proceedings of the National Academy of Sciences USA* 101: 18042–18047.

Selva, N. (2004) *The role of scavenging in the predator community of Białowieża Primeval Forest (E Poland)*. PhD Thesis. University of Sevilla. Sevilla.

Selva, N. and Fortuna, M. A. (2007) The nested structure of a scavenger community. *Proceedings of the Royal Society of Biological Sciences* 274: 1101-1108.

SEO/BirdLife (2010) *Estado de conservación de las aves en España en 2010*. SEO/BirdLife. Madrid.

SEO/BirdLife (2012) *Correcto uso de productos rodenticidas en espacios abiertos." Acciones para la lucha contra el uso ilegal de veneno en el medio natural en España*" (LIFE 08 NAT/E/000062). SEO/BirdLife. Madrid.

Sergio, F., Blas, J., Forero, M., Fernandez, N., Donázar, J.A. and Hiraldo, F. (2005) Preservation of wide-ranging top predators by site-protection: Black and red kites in Doñana National Park. *Biological Conservation* 125: 11-21.

Shaffer, M. L. (1990) Population Viability Analysis. *Conservation Biology* 4: 39-40.

Simón, M. A. et al. (2007) The reintroduction of the Bearded Vulture *Gypaetus barbatus* in Andalusia, southern Spain. *Vulture News* 56: 29-40.

Simón, M. A. et al. (2012) Reverse of the Decline of the Endangered Iberian Lynx. *Conservation Biology* 26: 731-736.

Skartsi, T., Elorriaga, J., Vasilakis, D. and Poizaridis, K. (2008) Population size, breeding rates and conservation status of Eurasian black vulture in the Dadia National Park, Thrace, NE Greece. *Journal of Natural History* 42: 345–353.

Sklyarenko, S. and Katzner, T. (2012) The Black vulture *Aegypius monachus* in Central Asia. In: Dobado, P. M. and Arenas, R. (cords.) *The Black Vulture: Status, Conservation and Studies*. Junta de Andalucía. Córdoba.

Soulé, M. E. (1985) What is Conservation Biology? *Bioscience* 35: 727-734.

Soulé, M. E. (1986) *Conservation Biology: the science of scarcity and diversity*. Sinauer Associates, Inc. Sunderland.

Soulé, M. E., Estes, J. A., Berger, J. and Del Rio, C. M. (2003) Ecological effectiveness: Conservation goals for interactive species. *Conservation Biology* 17: 1238–1250.

Soulé, M. E., Estes, J. A., Miller, B. and Honnold, D. L. (2005) Strongly interacting species: conservation policy, management and ethics. *Bioscience* 55: 168-176.

Speziale, K.L., Lambertucci, S.A. and Olsson, O. (2008) Disturbance from roads negatively affects Andean condor habitat use. Biological Conservation 141: 1765-1772.

Steidl, R.J. and Anthony, R.G. (2000) Experimental effects of human activity on breeding Bald Eagles. *Ecological Applications* 10: 258–268.

Suter, W., Graf, R.F. and Hess, R. (2002) Capercaillie *Tetrao urogallus* and avian biodiversity: testing the umbrella-species concept. *Conservation Biology* 16: 778–788.

Sutherland, W. J., Pullin, A. W., Dolman, P.M. and Knight, T. M. (2004) The need for evidence-based conservation. *Trends in Ecology and Evolution* 19: 305–308.

Sutherland, W. J. et al. (2010) Standards for documenting and monitoring bird reintroduction projects. *Conservation Letters* 3: 229–235.

Taggart, M. A., Reguero, M. M., Camarero, P. R. and Mateo, R. (2011) Should legislation regarding maximum Pb and Cd levels in human food also cover large game meat? *Environmental International* 37: 18-25.

Tella, J. L. (2001) Action is needed now, or BSE crisis could wipe out endangered birds of prey. *Nature* 410: 408.

Tellería, J. L. (2009a) Overlap between wind power plants and Griffon Vultures *Gyps fulvus* in Spain. *Bird Study* 56: 268-271.

Tellería, J. L. (2009b) Potential effects of wind farms on migratory birds in Spain. *Bird Conservation International* 19: 131-136.

Terrasse, M., Sarrazin, F., Choisy, J. P., Clemente, C., Henriquet, S., Lecuyer, P., Pinna, J. L. and Tessier, C. (2004) A success story: the reintroduction of Eurasian griffon *Gyps fulvus* and black *Aegypius monachus* vultures to France. In: Chancellor, R. D. and Meyburg, B. U. (Eds.). *Raptors worldwide: proceedings of the VI World Conference on Birds of Prey and Owls*. World Working Group on Birds of Prey and Owls-MME/Birdlife. Budapest-Berlin. pp. 127-145.

Tewes, E. (1996). *The European black vulture Aegypius monachus, management techniques and habitat requirements*. PhD Thesis. University of Wien.

Thompson, J. D. (2005) *Plant evolution in the Mediterranean*. Oxford University Press. Oxford.

Tscharntke, T. et al. (2012) Landscape moderation of biodiversity patterns and processes - eight hypotheses. *Biological Reviews* 87: 661-685.

Traverso, J. M. (2001). Nidificaciones sobre árbol de buitre leonado en España. *Quercus* 180: 23-25.

Valverde, J. A. (1966) Sobre buitres negros en Andalucía. *Ardeola* 12: 101-115.

Vasilakis, D., Poizaridis, K. and Elorriaga, J. (2006). Breeding season range use of a Eurasian Black Vulture *Aegypius monachus* population in Dadia National Park and the adjacent areas, Thrace, NE Greece. In: Houston, D. C. and Piper, S. E. (Eds.) *Proceedings of the International Conference on Conservation and Management ofVulture Populations*. Natural HistoryMuseum of Crete -WWF Greece. Thessaloniki.

Velando, A. and Freire, J. (2001) How general is the central-periphery distribution among seabird colonies? Nest-spatial pattern in the European Shag. *Condor* 103:544-554.

Virani, M. Z., Kendall, C., Njoroge, P. and Thomsett, S. (2011) Major declines in the abundance of vultures and other scavenging raptors in and around the Masai Mara ecosystem, Kenya. *Biological Conservation* 144: 746-752.

Vitousek, P. M. (1994) Beyond global warming: ecology and global change. *Ecology* 75: 1861-1876.

Walston, J. et al. (2010) Bringing the Tiger Back from the Brink—The Six Percent Solution. *PLoS Biology* 8: e1000485.

Wenny, D. G., DeVault, T. L., Johnson, M. D., Kelly, D., Sekercioglu, C. H., Tomback, D. F. and Whelan, C. J. (2011) The need to quantify ecosystem services provided by birds. *The Auk* 128: 1-14.

Whelan, C. J., Wenny, D. G. and Marquis, R. J. (2008) Ecosystems Services Provided by Birds. *Annals of the New York Academy of Sciences* 1134: 25-60.

Wilkin, T.A., King, L.E. and Sheldon, B. C. (2009) Habitat quality, nestling diet, and provisioning behaviour in great tits *Parus major*. *Journal of Avian Biology* 40:135-145.

Wilson, E. O. (1992) *The diversity of Life*. Belknap Press. Cambridge.

Winemiller, K. O. (1992). Life-history strategies and the effectiveness of sexual selection. *Oikos* 63: 318-327.

WWF España (2011) *La lucha contra el veneno en España. Clasificación por Comunidades Autónomas*. WWF España. Madrid.

Young, J., Watt, A., Nowicki, P., Alard, D., Clitherow, J., Henle, K., Johnson, R., Laczko, E., McCracken, D., Matouch, S., Niemela, J. and Richards, C. (2005) Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. *Biodiversity and Conservation* 14: 1641–1671.

Zhang, L., Wang, M. H., Hu, J. and Ho, Y. S. (2010) A review of published wetland research, 1991-2008: ecological engineering and ecosystem restoration. *Ecological Engineering* 36: 973-980.

Zink, R. M. (2004) The role of subspecies in obscuring avian biological diversity and misleading conservation policy. *Proceedings of the Royal Society B: Biological Sciences* 271: 561-564.

Zisenis, M. (2009) To which extent is the interdisciplinary evaluation approach of the CBD reflected in European and international biodiversity-related regulations? *Biodiversity and Conservation* 18: 639-648.

Zuberogoitia, I., Martínez, J.E., Margalida, A., Gómez, I., Azkona, A. and Martínez, J.A. (2010) Reduced food availability induces behavioural changes in Griffon vulture. *Ornis Fennica* 87: 52-60.

Zuberogoitia, I., Zabala, J., Martínez, J. A., Martínez, J. E. and Azkona, A. (2008) Effect of human activities on Egyptian vulture breeding success. *Animal Conservation* 11: 313–320.