Di Vittorio M., 2006. Reintroduction of the Griffon Vulture Gyps fulvus in Nebrodi Regional Park, Sicily. In: Houston, D.C...

Massimiliano Di Vittorio
Ecologia Applicata Italia srl

PROCEEDINGS OF THE INTERNATIONAL CONFERENCE ON

CONSERVATION AND MANAGEMENT OF VULTURE POPULATIONS

14-16 NOVEMBER 2005
THESSALONIKI, GREECE

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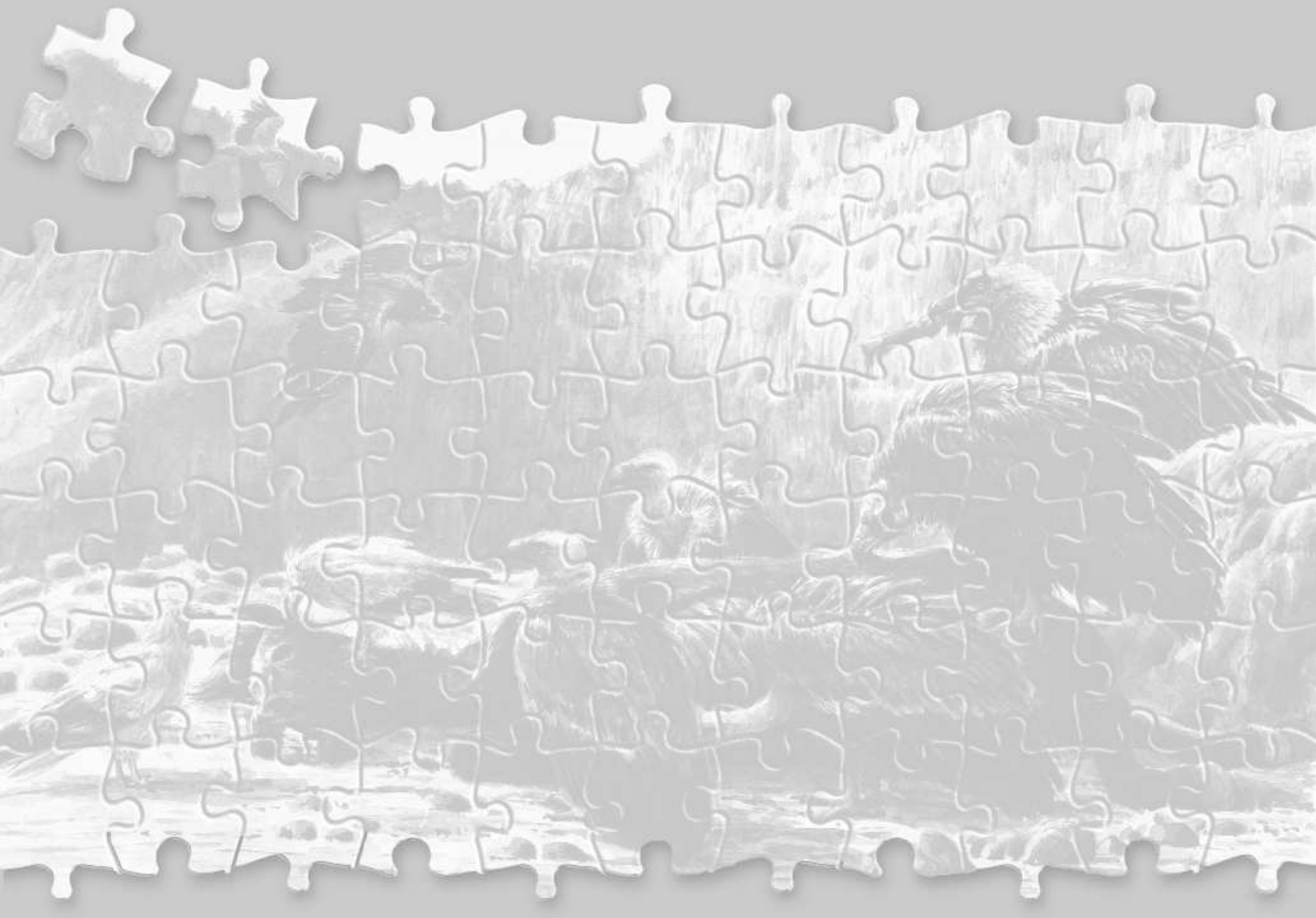

14-16 NOVEMBER 2005
THESSALONIKI, GREECE

WWF

PREFECTURE OF EVROS

EUROPEAN COMMISSION DG ENVIRONMENT

NATURA 2000

REGION OF CRETE FORESTRY DEPARTMENT

NATURAL HISTORY MUSEUM OF CRETE UNIVERSITY OF CRETE

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For further information:
University of Crete – Natural History Museum of Crete
Knossos Avenue, P.O. Box 2208, GR-71409 Heraklion, Crete, Greece
Tel./Fax +30 2810 324366, E-mail: mprobonas@nhmc.uoc.gr
URL: http://www.nhmc.uoc.gr

WWF Greece, 26, Filellinon street, GR-10558 Athens, Greece
Tel. +30 210 3314893, Fax +30 210 3247578, E-mail: c.liarikos@wwf.gr
URL: http://www.wwf.gr

Produced by: "ANIMA GRAPHICS" Fragini 9, 546 24 Thessaloniki, Tel.:: +30 2310 277997

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EDITORIAL

The International Conference on the "Conservation and Management of Vulture Populations" (14-16 November 2005, Thessaloniki, Greece) was co-organised by WWF Greece and the Natural History Museum of the University of Crete (NHMC). It was co-funded by the European Commission in the framework of the two LIFE-Nature projects that the organisers were implementing, namely "Protection of Birds of Prey in the Dadia Forest Reserve" (LIFE02 NAT/GR/8497) and "Conservation Actions for *Gypaetus barbatus* and Biodiversity in Crete" (LIFE02 NAT/GR/8492), respectively.

The conference proved quite successful, since more than 150 experts, scientists, NGO representatives, students and other interested individuals participated. Emphasis was given to transnational cooperation and representatives of Balkan NGOs (Bulgaria, Croatia, FYROM, Serbia and Montenegro) as well as Turkey, Ukraine and Israel were invited to participate. Invited speakers from Europe and other parts of the world presented their experience regarding the topics of the conference (e.g. monitoring systems, artificial feeding, habitat modelling, population dynamics, reintroduction projects, conservation genetics, and specific studies of LIFE projects), while an extensive discussion took part after the end of the oral presentations. A parallel poster exhibition was also held.

We truly believe that the present volume will prove to be of great help to relevant scientists, NGOs, policy-makers and managers working in the field. In this framework we would like to dearly thank Prof. David C. Houston and Prof. Steven E. Piper, who spent their valuable time in an effort to edit all papers and have this volume ready for publication in time, and, of course, all those distinguished scientists and practitioners who offered to contribute to this collective work.

The abstracts of presentations that were not sent for publication can be found in Annex I, while all abstracts of posters are included in Annex II. Although Dr Antoni Margalida did not manage to participate at the conference, he has sent an interesting paper for publication.

Dr Michalis Probonas
Physicist – Environmental Scientist
Coordinator of Public Awareness Actions
LIFE Project, NHMC

Dr Giorgos Catsadorakis
Scientific Advisor
WWF Greece

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THE ORGANISERS OF THE CONFERENCE

The conference was co-organised by WWF Greece and the Natural History Museum of the University of Crete (NHMC), and was co-funded by the European Commission. The organisation fell under the two LIFE-Nature projects that the partners were implementing and which referred to the "Protection of Birds of Prey in the Dadia Forest Reserve" (LIFE02 NAT/GR/8497) and to the "Conservation Actions for *Gypaetus barbatus* and Biodiversity in Crete" (LIFE02 NAT/GR/8492), respectively.

WWF GREECE

WWF Greece was founded in 1990 and is based in Athens, Greece. It belongs to the international family of WWF (World Wide Fund for Nature) but it constitutes an independent and self-financed national organisation, dedicated to the protection of Greek natural heritage. Towards that aim, WWF Greece deals with a variety of issues, like forest protection, use of toxic substances, water management, energy, rural development, etc. Furthermore, it organises information and sensitisation campaigns at the national and European level.

WWF Greece also runs many local action units, working on the conservation and sustainable development of environmentally important Greek areas. In this way it contributes to our vision for a development of the periphery that respects our natural heritage and promotes harmony between man and nature. From the list of local conservation units, we can mention the teams of Dadia, North Pindus, Kastelorizo, Lake Kerkini, Zakynthos and Prespa, but also older programmes like those of the Evros Delta, the Axios Delta, Nestos, Amvrakikos gulf, etc.

For the successful implementation of its actions, WWF Greece is supported by more than 10,000 members and a wide nexus of volunteers.

For more information on WWF Greece, please see: **http://www.wwf.gr** (pages in Greek).

NATURAL HISTORY MUSEUM OF CRETE

The Natural History Museum of Crete (NHMC) was established in the early '80s. It is part of the Faculty of Biological Science of the University of Crete. It consists of five departments: Zoology, Botany, Geology – Palaeontology, Mineralogy and Anthropology. It is supervised by the Dean of the Faculty together with the Presidents of the Biological and Geological Departments. The Museum is directed by a five-member Council, which consists of the directors of its corresponding departments.

The aim of the Natural History Museum of Crete is to be established as the main research institute for the study of the natural environment of Eastern Mediterranean and maintain its role in the education and civil awareness. This goal will be achieved through the following actions:

- Implementation of basic and applied research in the protection and conservation of the natural environment, its biodiversity and the endemic or rare forms of life.

- Management and conservation of the Greek natural ecosystems and especially those that are threatened by human impact.
- Development of complete animal and plant collections.
- Education of undergraduate and postgraduate students, scientific supervision of research projects and support of the environmental education in schools.
- Public awareness on matters of environmental conservation through: a) temporary and permanent exhibitions all over Crete, b) seminars addressed to public, and c) series of publications regarding the environment.

For more information on NHMC, please see: **http://www.nhmc.uoc.gr**

PRESENTATION OF LIFE – NATURE PROJECTS

CONSERVATION OF BIRDS OF PREY IN THE DADIA FOREST RESERVE

The Dadia Forest Reserve rests in the Prefecture of Evros, Region of Eastern Macedonia and Thrace, very close to the Greek borders with Bulgaria and Turkey. It surrounds the Soufli and Tychero municipalities, with a total population of 11,000 – mainly involved in stockbreeding, agriculture, forestry and ecotourism.

The Dadia Forest Reserve constitutes a characteristic Mediterranean landscape, which has been shaped through many centuries of man-nature coexistence and interaction. Vegetation is mainly composed from mature Calabrian Pine *Pinus brutia* and Black Pine *Pinus nigra* forests and from oak forests. The dense forest cover is frequently disrupted by openings, small meadows and farmed areas, resulting to a mosaic coverage ideal for birds of prey species.

The National Park hosts 36 out of the 38 diurnal raptor species of Europe, among which some very important and rare species like the Imperial Eagle *Aquila heliaca* or the Lesser Spotted Eagle *Aquila pomarina*. It is also one of the few European areas that host so many different raptor species among them three different vulture species (Black Vulture, Egyptian Vulture and Griffon Vulture). Among all of the hosted raptor species, the flag species of the park is the Black Vulture *Aegypius monachus*, which preserves in the forest its last breeding colony of the Balkans – and one of the few remaining in Europe. Its population in Dadia exceeds 90-100 individuals, including around 20-22 breeding pairs.

The LIFE project under the title: "Conservation of Birds of Prey in the Dadia Forest Reserve, Greece", was approved by the European Commission in 2002. Through a series of actions, the project targets the implementation of an integrated plan for the scientific study and the conservation of the site. In brief, the actions of the project are the following:

Black Vulture Management Plan: The conservation and management of birds of prey – and of any other species – presupposes extensive research and proper action designs. For that reason, one of the first actions of the programme referred to the preparation of a management plan which would concentrate all existent knowledge concerning the black vulture population of the area, it would describe the present situation and would propose measures for the short-, medium- and long-term conservation of the species. The "management plan", which has been already completed, represents the "backbone" of the programme, as it guides the application and evaluation of the majority of other actions.

Monitoring of the Forest Reserve and of the Black Vulture: Any conservation attempt should be accompanied by the application of an integrated monitoring scheme, which can facilitate the evaluation of results and the confinement of problems. The monitoring of the protected area and its residing species is based on an integrated monitoring plan, which was prepared by WWF

Greece during the year 2000, and which combines the monitoring and recording of indices related to the forest's ecology and the birds of prey population. Especially for the Black Vulture population, the LIFE programme finances the use of state-of-the-art methodologies, like satellite tracking.

Maintenance of Forest Openings: One of the most important characteristics of the Dadia Forest Reserve is to be found in its mosaic coverage, which combines forested areas with forest openings, meadows and farmed land. This mosaic allows the birds on the one hand to find cover and nesting areas and on the other to enjoy open fields where they can hunt. The conservation of this valuable mosaic is undertaken by the programme, through the maintenance of forest openings, which have "closed" due to the reduction of both domestic and wild herbivorous.

Creation of Small Artificial Ponds: Complementary to the previous action, the creation of small artificial ponds, through the construction of "micro-dams", will improve water availability in the strictly protected area (especially during the summer months) and will create small biotopes for reptiles – which constitute a major food source for raptors.

Creation of New Feeding Sites: The creation of the Dadia feeding site has been an action catalytic for the survival and conservation of the vulture species. Nowadays, circumstances regarding the availability of carcasses in the forest have not improved and for that reason the transfer of dead animals in the forest is still a measure vital for the conservation of vultures. Yet, supply should be undertaken in away that could better simulate natural conditions and should be delivered in more than one place. The new feeding sites target strictly conservation purposes and are not combined with the attraction of tourists as the present one is.

Information / Awareness: Dadia's local community has consistently supported WWF Greece actions in the area and has conduced to the successful completion of its projects. Through a multitude of informative actions, the present programme wishes to keep-up with this tradition of good cooperation, by keeping the local community and the visitors informed about the project's developments.

For more information on the LIFE project, please see:

http://www.wwf.gr/index.php?option=com_content&task=view&id=87&Itemid=133

CONSERVATION ACTIONS FOR THE BEARDED VULTURE AND BIODIVERSITY IN CRETE

The Bearded Vulture *Gypaetus barbatus* is considered as one of the rarest raptors in both Greece and Balkans, since its breeding population can be found only in Crete and the relevant number of breeding pairs of the species is only four. In 1998, DG Environment of the European Commission funded a LIFE – Nature project on the "Conservation of the Bearded Vulture in Greece" (B4-3200/98/444), which was implemented by the Natural History Museum of Crete (NHMC) and the Hellenic Ornithological Society (HOS) during the period October 1998 – February 2002.

In the framework of the same funding measure of the European Commission (LIFE – Nature 2002), the Natural History Museum of Crete, in collaboration with the Forestry Department of the Region of Crete and the Municipality of Inachorio, undertook the implementation of a new project on "Conservation Actions for *Gypaetus barbatus* and Biodiversity in Crete" (LIFE02 NAT/GR/8492). The duration of the project is four years and its implementation started on July

2002. The main objectives of the aforementioned project are the implementation of the most urgent conservation actions for the species in Crete and the elaboration of specific conservation measures in mountainous areas of Crete.

The project aims to the conservation of the current population of the Bearded Vulture *Gypaetus barbatus* in Crete, as well as the conservation of the biodiversity of the island, through the confrontation of specific human threats to wildlife (e.g. direct execution and use of poisons, low food availability, desertification of ecosystems and habitat degradation etc.). In addition, the project aims to the environment-friendly development of rural areas, through the promotion of ecotourism and local products at the project sites.

Apart from the conservation of the Bearded Vulture population, the project focuses on the conservation of Crete's biodiversity. Through the implementation of certain actions, species such as the Griffon Vulture *Gyps fulvus*, the Golden Eagle *Aquila chrysaetos*, the Bonelli's Eagle *Hieraaetus fasciatus*, the Peregrine *Falco peregrinus* and the Lanner *Falco biarmicus*, which are also protected under Directive 79/409/EEC, are expected to benefit significantly from the project.

The main actions of the project are the following:

- Intensive monitoring of the Bearded Vulture population and the population of other protected birds of prey at the mountain ranges of Crete. The monitoring is accomplished by annual censuses of the breeding pairs, observations of marked / ringed individuals, as well as tracking of radio-tagged birds.
- Rescue of the second Bearded Vulture chick (which never survives in the nest), which will be raised in captivity by foster parents in European Breeding Centres. The ultimate goal of this action is the creation of a breeding stock and the release of its offsprings back on the island. The action is considered to be absolutely necessary, due to the critical status of the species population. The monitoring of breeding stages (e.g. incubation, hutching etc.) is accomplished with specific micro-cameras, which have been placed in two nests.
- Artificial feeding through the provision of supplementary food at 6 mountainous areas of Crete.
- Design and implementation of an effective warding scheme in the Wild Life Reserves of mountainous Crete from relevant Forest Services and Hunting Associations. Wardens of Crete will attend relevant seminars for improving their specific knowledge on raptors' biology, observation and warding. The seminars will be organised by the Natural History Museum of Crete (NHMC).
- Delivery of special wasp-traps to beekeepers of Crete, aiming to reduce the use of poisoned baits for wasps. In general, the use of poisoned baits is considered to be an extremely detrimental action for birds of prey.
- Management of indigenous game species. The action includes the breeding of Chukar partridge of local genetic origin and the function of adaptation aviaries, aiming at the enrichment of the natural populations of game species at the Wild Life Reserves of Crete.
- Restoration of degrading areas at a pilot basis (e.g. through the planting of indigenous species), construction of relevant infrastructure for the support of extensive pastoralism (e.g. waterers) and construction of small ponds for wildlife use are some of the important conservation actions under implementation.
- Promotion of environment-friendly agricultural and pastoral practices through a relevant wide campaign for agri-environment regulations of the European Union, and also support for the verification and promotion of local biological products.

- Promotion of ecotourism in the mountainous areas of the project, which will be based to the particular value of the natural and human environment. The action includes the restoration and signing of old mountainous trails, the construction of bird observatories, the establishment of Information Centres, the organisation of exhibitions and fests, and the production of relevant information material (e.g. ecotouristic guides). The production of relevant material for Environmental Education and the delivery of videotapes will empower the relevant public awareness campaign in Crete.

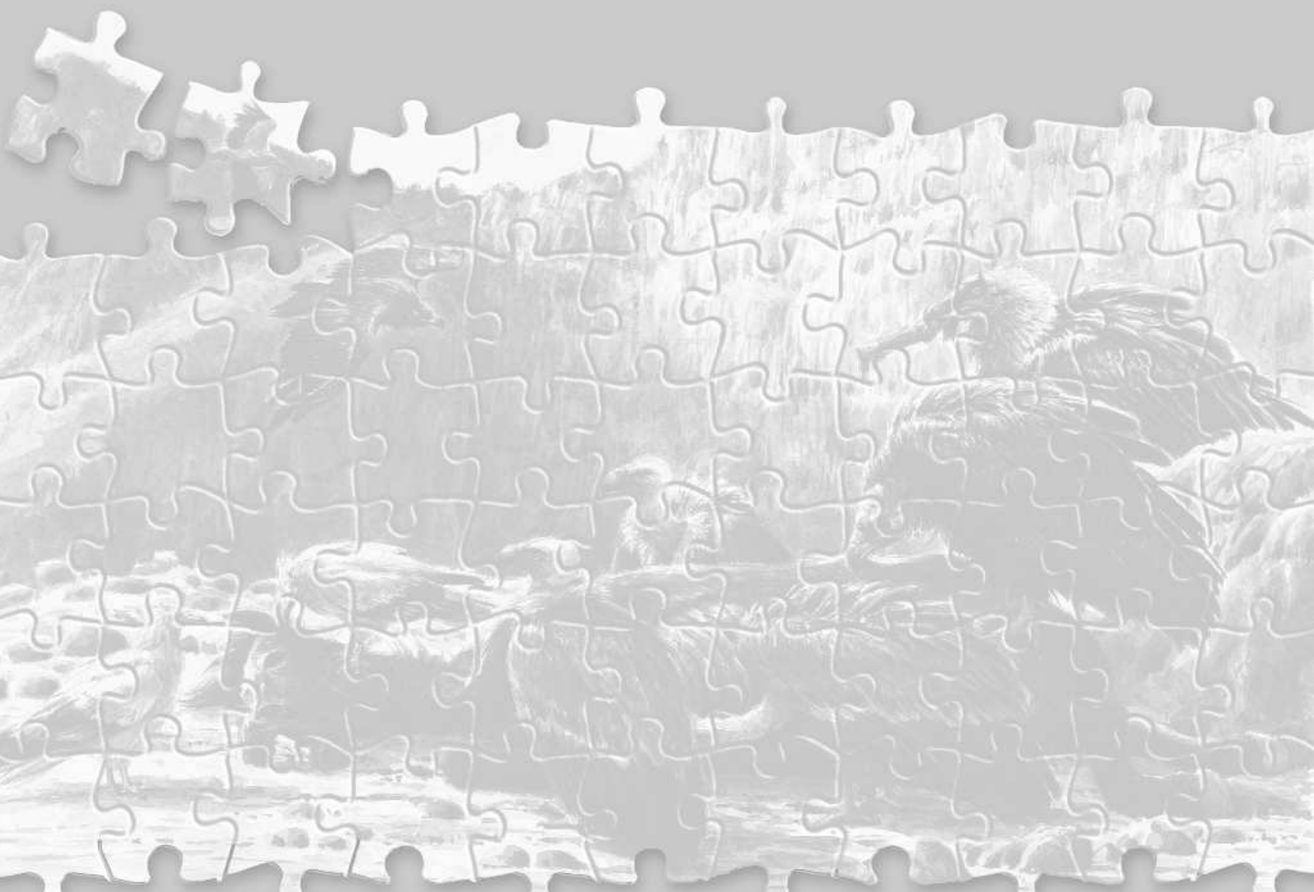
For more information on the LIFE project, please see: **http://www.nhmc.uoc.gr/life_gypaetus/**

PROGRAMME OF THE CONFERENCE

DATE	TIME	LECTURES
MONDAY, 14/11/2005		
MORNING		
	08:30-09:30	**REGISTRATION**
	09:30-10:30	**OPENING SPEECHES**
SESSION: **MONITORING SYSTEMS**	10:30-10:50	**The Design and Implementation of Telemetry Studies: Applications in Vulture Species** **Dr. OFER BAHAT**, Department of Environmental Science and Chemistry, University of Indianapolis, Mar-Elias Campus, Ibillin, Israel
	10:50-11:10	**Possibilities and Limitations of Biotelemetry and Radio-tracking Techniques in Vultures Species** **Dr. RALF BOEGEL**, EGVWG, Germany
	11:10-11:30	**Effect of Monitoring Frequency and Date on Estimates of Abundance and Productivity of Colonial Black Vultures in Spain** **JAVIER DE LA PUENTE**, SEO/BirdLife, Spain
	11:30-12:00	**COFFEE BREAK**
SESSION: **ARTIFICIAL FEEDING**	12:00-12:30	**Supplementary Feeding Programs: How necessary are they for the maintenance of numerous and healthy vultures populations** **Prof. STEVE PIPER**, Ornithological Support Service, South Africa
	12:30-12:50	**Food Exploitation by Griffon Vultures: The effect of vulture restaurants in Spain** **ALVARO CAMIÑA**, ACRENA, Spain
	12:50-14:00	**DISCUSSION**
AFTERNOON		
SESSION: **HABITAT MODELLING**	16:00-16:20	**Ecological Requirements of Reintroduced Species and the Implications for Release Policy: The case of the Bearded Vulture** **Dr. RAPHAËL ARLETTAZ**, University of Bern, Switzerland
	16:20-16:40	**Modelling Vulture Habitats in the Caucasus** **MICHAEL McGRADY**, Natural Research, Ltd., Austria
	16:40-17:00	**Habitat Modelling for Black Vulture *Aegypius monachus* L. in Cabañeros National Park** **JOSÉ JIMÉNEZ GARCÍA-HERRERA**, Director of Cabañeros National Park, Spain
	17:00-17:20	**Habitat Modelling of the Bearded Vulture: Feasibility study for the Andalucian reintroduction project** **JOSÉ M. PADIAL**, Fundación Gypaetus, Spain
SESSION: **POPULATION DYNAMICS**	17:20-17:40	**Dynamic of Restored Populations of Griffon Vultures in Southern France** **Dr. FRANCOIS SARRAZIN**, Lab. Conservation des Espèces, Restauration et Suivi des Populations, France
	17:40-18:00	**Population Viability Analysis of Eurasian Griffon in Croatia** **GORDANA PAVOKOVIC**, Eco-center "*Caput Insulae*-Beli", Croatia
	18:00-19:00	**DISCUSSION**
TUESDAY, 15/11/2005		
MORNING		
SESSION: **REINTRODUCTION PROJECTS**	09:30-10:00	**Reintroduction Programmes for Vultures** **Prof. DAVID C. HOUSTON**, Institute of Biomedical and Life Sciences, Glasgow University, UK
	10:00-10:20	**Reintroduction of Black Vulture in Mallorca, Spain** **Dr. EVELYN TEWES**, BVCF, Spain

DATE	TIME	LECTURES
TUESDAY, 15/11/2005		
MORNING (continued)		
SESSION (continued): **REINTRODUCTION PROJECTS**	10:20-10:40	**Long-Term Reintroduction Projects of Griffon and Black Vultures in France** **MICHEL TERRASSE**, Ligue Française pour la Protection des Oiseaux, France
	10:40-11:00	**Actions for the Reintroduction of the Bearded Vulture (*Gypaetus barbatus*) in Andalusia** **SERGIO COUTO**, Fundación Gypaetus, Spain
	11:00-11:20	**The Status of Griffon Vulture in Italy** **FULVIO GENERO**, Riserva Naturale del Lago di Cornino, Italy
	11:20-12:00	**COFFEE BREAK**
SESSION: **CONSERVATION GENETICS**	12:00-12:20	**Genetics of restored populations of Griffon Vulture in France and Europe** **PASCALINE LE GOUAR**, Research Unity of "Species Conservation, Restoration and Monitoring of Populations" (MNHN/CNRS/UPMC), Paris, France
	12:20-12:40	**Using Microsatellite Markers to Infer the Genetic Structure of *Aegypius monachus* (Aves: Accipitridae)** **ARIS PARMAKELIS**, University of Crete – Natural History Museum of Crete, Greece
	12:40-13:00	**The Genetic Structure of the Bearded Vulture (*Gypaetus barbatus*) Population in Crete** **ARIS PARMAKELIS**, University of Crete – Natural History Museum of Crete, Greece
	13:00-14:00	**DISCUSSION**
AFTERNOON		
SESSION: **LIFE PROJECTS: SPECIFIC STUDIES**	16:00-16:20	**Radio-Telemetry of a Black Vulture *(Aegypius monachus)* Population in the Dadia National Park and Adjacent Regions: Methodology and Preliminary Results** **DIMITRIS VASILAKIS**, WWF Greece-Dadia Project, Greece
	16:20-16:40	**Discrimination of the Sex in the Cinereous Vulture *Aegypius monachus* Using Morphometric Techniques** **JAVIER ELORRIAGA**, WWF Greece-Dadia Project, Greece
	16:40-17:00	**Impact of Wind Farms on Birds in Evros and Rhodopi, Greece: Preliminary results** **CARLOS RUIZ**, WWF Greece-Dadia Project, Greece
	17:00-17:20	**Evaluation of the Use of Mini-Cameras in Nest Monitoring of the Bearded Vulture in Crete** **COSTAS GRIVAS**, University of Crete – Natural History Museum of Crete, Greece
	17:20-18:30	**DISCUSSION – CONCLUSIONS**
WEDNESDAY, 16/11/2005		
MORNING		
	09:30-13:30	**5th Annual Meeting of the European Griffon Vulture Working Group (EGVWG)**
	16:00	Departure for the excursion to National Park of Dadia-Lefkimi-Soufli Forest, Evros Prefecture

PAPERS BASED
ON ORAL PRESENTATIONS



POSSIBILITIES AND LIMITATIONS OF BIOTELEMETRY AND RADIO-TRACKING TECHNIQUES IN VULTURE SPECIES*

Ralf Bögel

Dauchingerstr. 64, 78078 Niedereschach, Germany
E-mail: Ralf.Boegel@t-online.de

Abstract

From the beginnings of wildlife and bird biotelemetry until now, an enormous development has taken place. Nowadays, a wide range of telemetry techniques is available which require a precise definition of the dominant questions to be answered, a careful evaluation of the advantages and disadvantages of various techniques, the selection of the most appropriate material with regard to the budget available and a rigorous study design and implementation. One false estimation in this chain of judgments may challenge the outcome of a study and can even affect the reputation of a whole project. Furthermore, the techniques of data plausibilisation and analysis contribute substantially to the scientific output of the study and will determine to what extent the study objectives are being met. Therefore, these methodological aspects are of utmost importance and deserve as much attention as biological, ecological, conservational or educational aspects.

This paper focuses on the involved technical aspects, discusses the advantages and drawbacks of certain techniques with special reference to their applicability under field conditions and the suitability for monitoring of vulture species. General recommendations and guidelines are given.

Keywords: Telemetry; VHF; ARGOS; GPS; Study design; Vultures.

****Publisher's note:*** The present paper was received some weeks overdue and the editors of the volume did not have the opportunity to comment upon it. It is thus included here only with minor linguistic corrections.

Introduction and Definition of Problems

From the beginnings of wildlife and bird radio-telemetry in the early 1950s until now, an enormous technical development has taken place. Nowadays, a wide range of telemetry techniques is available which include manual tracking with VHF- and UHF telemetry, automatic data logging systems with presence/absence recording or automatic location finding, satellite telemetry for global position determination using the ARGOS satellite system and highly accurate GPS techniques. In the field of bio-telemetric applications, a wide range of sensors for acquiring additional eco-physiological parameters has been applied (Bögel, 1996, 1999). Concerning data transmission, also various technical approaches have evolved ranging from analogue over simple digital to complex modulation schemes. In some recent designs, a store-on-board logging procedure is implemented, either to reduce weight or in order to increase data safety. While tracking and location finding is commonly the main interest in projects with a predominantly conservation or ecological focus, generally studies with an eco-physiologic focus have mainly been restricted to controlled conditions. However, some projects have proved that the study of eco-physiologic questions can well go along with the applicability under field conditions, and that such studies may be worthwhile especially under natural environmental conditions (Bögel, 1999, 2001; Prinzinger *et al*. 2002).

Besides the more technical aspects, the sociability of transmitter fixation can be a highly sensitive subject. Especially in high-image conservation projects where the welfare of individual specimens is certainly of utmost importance, this question has been the subject of highly controversial and often emotional discussions. Again, a couple of methods have been developed for transmitter fixation, each with specific pros and cons. The same holds true for the appropriate trapping techniques, which are generally a prerequisite for wildlife-tagging.

If not just a simple monitoring of the tagged birds is of primary interest and the data are intended to be used for some kind of scientific analysis, a whole set of additional methodological considerations require careful attention. In this context, the following points are of crucial importance:

1) Which are the main study objectives? Will the application of radio-telemetry deliver a significant contribution to the posed questions or are there any alternative methods available?
2) Which are the minimum sample sizes, that statistically allow for meaningful results (considering sex and age classes under study)? Is there a realistic likelihood to obtain these sample sizes under field conditions (efficiency of trapping methods)?
3) What is the adequate study design in order to maximise the chance for obtaining the pursued results (species distribution, selection of study sites, logistic and financial constraints)?
4) What are the appropriate analysis methods in order to answer the addressed study objectives and scientific questions? Do these methods imply certain statistical assumptions and thus interact with the study design? Are the required resources (personal, expertise, technical, financial) available to the project?

Telemetry is definitely a "sexy" technique to use but it is quite obvious that if the above-mentioned points are not clearly defined and evaluated, its application is poorly justified. In general, the output of a project will be highly restricted if, initially, the data are being sampled and, retrospectively, experts are consulted and are asked to extract the "maximum" of the available data. Unfortunately, this is not an uncommon practise.

Methodological Aspects

It is definitely impossible to discuss thoroughly all relevant aspects within the framework of this paper. Instead, I will focus here on the subjects which are more related to trapping, radio-tagging and the applicability of the available techniques for vulture monitoring under field conditions. However, this does not imply that the other aspects are assessed to be of less importance. Concerning study design, statistical aspects and data analyses, I refer to standard literature (White & Garrot, 1990, Scheiner & Gurevitch, 1993, Bookhout, 1994; Bart *et al*. 1998; Kenward, 2001; Hennig & Bögel, 2004).

Concerning the selection of an appropriate specific radio-telemetry technique, the following questions deserve attention:

- What are the specific questions to be answered?
- Which sensors might deliver relevant information?
- Is a transmitter fixation with minimum risk possible?
- What are the pros and cons of available techniques?

In the following chapters, I address the related technical aspects in the order of their logical sequence within a project.

Trapping

Except in monitoring of released birds in a reintroduction or restocking project where the birds can be tagged in captivity, trapping is the first challenge to be addressed. Again, this is not the place to discuss trapping methods in detail. This is covered by other publications (e.g. Friends *et al*. 1994). However, I would like to review the techniques which have been successfully used in vultures. Basically, these include trapping with "entry-only volieries" where wild conspecifics are baited with meat and attracted by tame ones inside the cage (Iezekiel, Hatzofe, Woodly and Skartsi, pers. comm.), cannon nets (own experience; Slotta-Bachmayr and Genero, pers. comm.) and distant immobilization (Revers & Bögel, 1994). Working with a small semi-tame colony at Salzburg Zoo it was also possible to approach the vultures closely by baiting them with meat and catch them directly in an unattended moment (Bögel, 1996). It seems that each project may succeed best with an adaptation to the local situation. In this context it is important to note that a given trapping method which worked fine at the beginning of a project might loose efficiency over time. So be prepared to account for learning and habituation effects, which might undermine your project timetable.

Transmitter Fixation

There can be no doubt that any kind of tagging technique may impose some adverse effects with regard to behaviour, condition or mortality. Of course, this also holds true for radio-tagging techniques. Generally, the question of transmitter sociability and possible adverse impacts on the tagged animal has attracted comparably little scientific analysis, presumably due to the fact that the tagging period will exceed the duration of most studies. Kenward (2001) has analysed mortality rates in raptors derived from ringing and radio-tagging data (using tags with lifespans of several years and by analysing cohort dynamics data) and could not find a significant difference. So he concluded, that radio-tagging has no or a negligible impact on the bird survival rates (transmitters were fixed using tail-mounts and harnesses). Further studies have been made on the possible impact of a tag on the aerodynamics and flight performance, suggesting that this effect will usually be below 10% (see literature in Bögel, 1996).

There are various methods to fix a transmitter to a bird, ranging from glue-mounts, over tail- and leg-mounts, necklaces, harnesses to patagium mounts and implantation. All these techniques exhibit specific pros and cons (see Table 1). While glue-mounts are mainly used in small passerines, necklaces in grouse and leg-mounts in falconry for temporary tagging purposes, all other methods are commonly used in raptor species. When focusing on large raptors like vultures, tail-mounts will be the method with presumably the least impact (tag loss due to moulting is ensured) but with a rather unpredictable and generally short tagging period. It is also not suitable for fledglings (since tail feathers are still growing) and the maximum tag weight is clearly restricted due to the big distance from the centre of gravity and the maximum load of the tail feathers.

When relatively heavy or large transmitters have to be attached, it is beneficial to fix them as close to the centre of gravity as possible. This is one of the advantages of harness mounts. Other advantages are the long tagging period and the suitability for tagging fledglings if elastic harness material is being used. However, a controlled release mechanism is critical and uncontrolled breaking of a harness can definitely cause harm to the tagged bird (pers. obs., Kenward, 2001). Generally it is surprising, how commonly harness mounts are being used without implementing any kind of safe release mechanism such as a "weak-link" which could guarantee or at least greatly maximise the chance for a safe release of the tag. During the years, harness designs have evolved to a satisfying and safe alternative of transmitter fixation. If properly designed, they are one of the best options for transmitter fixation in raptors (Bögel, 1996; Kenward, 2001).

The Californian condor project has collected good experience with patagium tags (Wallace, pers. comm.) where some individuals are even tagged on both sides. However, I suggest that maximum transmitter load is more critical than with backpacks and that a certain risk of patagium traumas have to be accepted as observed in some cases even with wing tags attached to Griffon Vultures in a comparable way (Susic, pers. comm.).

In order to measure physiological parameters such as heart rate and body temperature, implanted tags have been applied and no adverse effects were observed (Walzer *et al.* 2000). Normally, implanted tags will result in a remarkably decreased system range due to the bad efficiency radius of implanted antennas. Using a transponder system where only the sensor is implanted and which communicates with the external transmitter through a wireless link, such drawbacks can be avoided (Bögel *et al.* 2002).

As with all studies and involved methods, a careful evaluation of possible risks and the likely scientific outcome is essential. This may also entail the discussion on trade-offs between the imposed risk for an individual specimen and the scientific, conservation and education benefit radio-telemetry can bring to the project.

As there are a couple of benefits linked to the use of sophisticated, large size transmitters (see below), harnesses are the preferred option in many cases. With a careful harness design, the imposed risk for a bird can be classified as very low. So, for most applications and study interests, I suggest that a harness fixation will

Table 1. Advantages and drawbacks of various techniques for transmitter fixation.
+ good, o moderate, - poor, -- extremely negative/demanding, o/- intermediate ranking.

	MOUNTING TECHNIQUE					
Criteria	Glue	Tail	Patagium	Leg	Harness	Implant
max. weight	-	o	o	o	+	o
centre of gravity	+	-	-	-	+	+
radio transmition	+	+	o	-	+	- -
tagging period	- -	-	o	+	+	+
use in juveniles	+	-	o	+	+	?
tag release	+	+	o	o / -	o / -	-
risk	+	+	o / -	o	o / -	o / -
experience required	+	+	o	+	-	- -

be the proper method for transmitter fixation in vulture monitoring. Considerable and positive experience is available on some dozens of tagged vultures with harness-mounted tags (own results, Bahat, Woodly, Hatzofe and McGrady, pers. comm.). In any case I would suggest 250 g (including harness material) as the maximum weight for large vultures such as Eurasian Griffon *Gyps fulvus* or European Black Vulture *Aegypius monachus*.

General Advantages and Drawbacks of Radio-Telemetry

As with every method, certain advantages and drawbacks are inherent with radio-telemetry. When comparing pros and cons (Table 2), it is quite obvious that the main advantage of radio-telemetry is generated by the systematic approach with increased efficiency of data sampling (monitoring intensity), the chance for a direct approach to the animal for visual observation and/or the provision of help (monitoring quality), and by the capability to collect sensor or activity data (which may not be possible through use of other techniques). However it must be recognised that even radio-telemetry cannot guarantee the timely provision of help, a fact illustrated by the case of “Antonio”, a rehabilitated vulture which was released in the “Hohe Tauern” region (Austria) and whose flight to the south ended with the bird drowned in the Mediterranean sea close to Genoa, Italy (Fig. 1). At least the fate of this individual specimen has been doc-

Table 2. Advantages and drawbacks of radio-telemetry.

Pros	Cons
Systematic and efficient data sampling	Requires efficient capturing methods
Analysis of spatial patterns and habitat use	High costs for equipement, labor and data-processing
Chance for taking action when animals require help (released bird)	Variable location quality (depending on method used)
Analysis of activity patterns (circadian and seasonal)	High effort of data evaluation in some automatic systems
Additional ecophysiologic information (sensors)	Risk of adverse effects

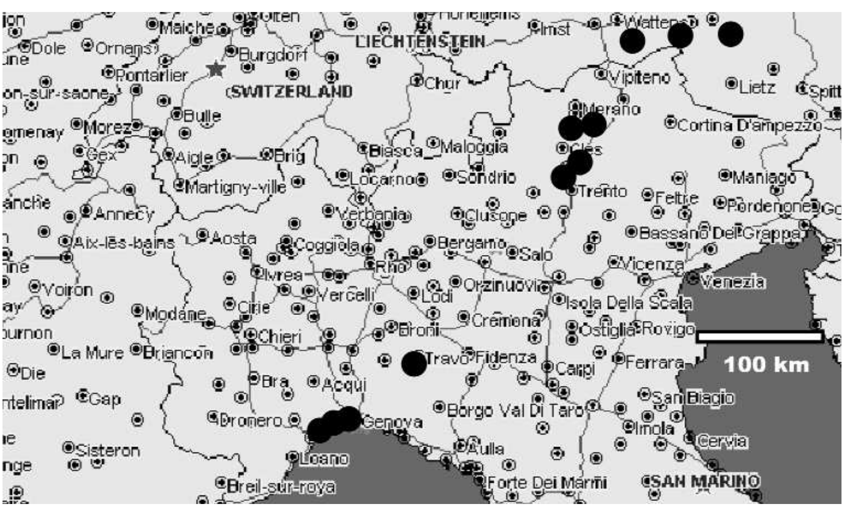


Figure 1. Monitored track of a rehabilitated Eurasian Griffon Vulture *Gyps fulvus* after release using a GPR-GSM tag (450 km travelled in 8 days with a total of 284 locations since release).

Table 3. Advantages and drawbacks of various tracking approaches.
+ good, o moderate, - poor, -- extremely negative/demanding, o/- intermediate ranking.

Criteria	manual tracking	ADF / ADL systems	ARGOS satellite	GPS GSM	GPS ARGOS
coverage	-	o	+	variable	+
data rates	variable	+	-	+	o
accuracy	variable	variable	-	+	+
plausibilisation	o	-	o	+	+
data access	variable	+	o	+	o / +
tracking compatibility	+	+	-	optional	optional
tagging period	+	+	o	o	-
manpower	- -	-	o	o	+
costs (personal ignored)	o	-	-	-	-
tag weight	+	+	o	-	-

umented. This can be helpful in optimising release techniques or adapting management plans.

It is important to note that some analysis techniques may presuppose independent data sets and thus require an adapted data sampling scheme. This has to be considered during study design. It is also helpful to evaluate the possible contribution of alternative methods. For example, systematic and coordinated ringing programs may reveal valuable information on roaming patterns.

Specific Techniques

It is stressed again that it is not possible to cover all telemetric approaches, which have been developed so far. I will concentrate on the most widely used and the most important ones, as well as those which are most promising for vulture monitoring. Table 3 gives an overview of the general advantages and drawbacks of various approaches.

Manual Tracking

Manual tracking is still the most frequent technique used in wildlife monitoring. Basically there are two different approaches of finding an animal's location: a) homing-in, and b) triangulation (Kenward, 2001). In triangulation simultaneous bearings are taken from different sites using directional antennas. The spatial resolution of this technique is basically a function of the directional characteristics of the antenna system (H-Adcock, yagi, stocked antenna array such as zero-peak configurations), the distance from the receiving station and the trigonometric configuration between the receiving sites and the animal's position. Angular resolutions vary from >15° (H-Adcock antenna) to < 1° (zero-peak antenna array with 10 element yagis). Unfortunately, the antenna designs which are best suited for mobile applications due to its small size (H-Adcock) have the poorest performance and vice versa. Some foldable yagi designs with approx. 5 elements may be the best compromise between size, directional performance and system range. Concerning the trigonometric configuration, best accuracies are obtained when bearings intersect in angles close to 90°. It is important to note that physical characteristics of the research area can impose additional causes of failure (Garrot *et al.* 1986).

In contrast to this, the homing-in procedure is a more pragmatic approach where the researcher just follows the bearing direction and approaches the animal by successively taking additional bearings and a consecutive approach towards the tagged bird until visual observation is possible or the signal is that strong that it can be assumed that the animal must be very close. So this is the technique generally used if manpower is restricted or a bird left his normal home range and the researcher tries to follow. However, reasonable location estimates before achieving visual observation are not possible as signal strength is an extremely poor indicator for the distance (at a given distance signal strength may vary as much as 1:100 due to the variation caused by different transmitter altitudes above ground and the antenna's orientation and polarisation).

Besides the low costs for receiver, antenna and transmitter, it is worth to mention that the requirements for transmitter power output are very low, allowing for high transmitter lifespans (up to 5 years) and low weight. Furthermore, many different sensor types are available allowing for a detection of activity, body orientation, air pressure / flight altitude, ambient or body temperature and various endogenous parameters like ECG or EMG (see Bögel, 1996). The main drawback of manual tracking is its very high labour intensity, especially if multiple specimens have to be monitored simultaneously.

Concerning practical issues, it is important to note that also the receiver being used has a significant influence on the system performance. Basically this is a result of physical characteristics such as sensitivity and robustness against interference caused by other strong signals within the frequency band. However some rather practical issues are often poorly solved, such as a sensitive signal strength indicator, an adequate characteristic of the gain control or a signal strength to audio frequency converter which greatly increases the ability of the human ear for accurately detecting the bearing direction. When working with numerous animals, some scanning ability makes tracking more comfortable and if PDM/PIM-modulated sensor data are being collected, an option for measuring the time characteristics of the received pulses might be required which is rarely available. Additionally, data-logging capabilities may be required (see below). Unfortunately, some processor-controlled receivers with sophisticated features are handicapped in the maximum system range by noise generated from their own CPU. When maximum range is an essential requirement, such receivers are probably not the best choice.

Automatic Stations

A couple of automatic, ground-based telemetry systems have been developed which are discussed in Bögel (1996) and Kenward (2001). These can basically be classified in presence/absence recordings and automatic direction and location finding systems. Some of these systems are compatible with standard VHF and UHF tags, other require special transmitters. Generally, such stations can deliver valuable results in local applications. However, due to the small reception range of each station (< 20 km), an adequate coverage of larger areas is impracticable due to the huge effort required for installing and maintaining such systems. Furthermore, cross-border applications may be difficult or impossible due to formal regulations like different frequency bands, which are applied by the authorities in different countries.

A severe handicap of such systems can also be linked to the plausibilisation of the generally huge amounts of logged data, which can turn out to be a real challenge. As information encoding in wildlife application is generally optimised for maximum range, low power consumption and maximum lifespan, data encoding techniques are generally not very robust against interference. Suitable algorithms for an automatic plausibilisation can be developed for some but not all related aspects. So the procedure of data processing can be really demanding and time-consuming.

Presence / Absence Recording and Data Logger Systems

They include simple presence / absence recordings at one or various places within the research area where the duration of presence is documented at specific spots, which may be roosting, nesting or feeding sites. However, the spatial resolution of such presence / absence recordings is poor and restricted to the reception range of a given equipment which may be adjusted to the study needs by varying the gain of the antenna and the receiver. When combining such stations with a telemetry transmitter equipped with sensors which code the activity status (motion sensors, orientation of body axis) or eco-physiological parameters of interest (air pressure / flight altitude, ambient or body temperature, heart rate, etc.), behavioural and physiological aspects can be documented with high temporal resolution within a given area (Bögel *et al.* 2000; Prinzinger *et al.* 2002). Accordingly, such stations are mainly used for studies where time budgets or eco-physiological questions are of primary importance. Suitable data-logging systems are available from Televilt/TVP, Lotek, ATS and others.

Automatic Direction (ADF) and Automatic Location Finding (ADL) Systems

Relatively few ground-based systems for wildlife applications have been developed which are capable of an automatic direction and location finding (Bögel 1996; Bögel et. al. 2001; Kenward, 2001). They can be distinguished in systems which determine the location of the transmitter by triangulation or by time of arrival measurements (TOA systems). In any case a minimum of 3 receiving stations is required to define the location of the animal (I exclude simple cross-bearings in this context because no plausibilisation check is possible). While time-of-arrival systems have a linear spatial resolution, the accuracy of locations varies greatly in systems which define a position through triangulation. In general, with triangulation, data quality is a function of distance from the receiving stations. One system which is capable to simultaneously track up to 100 animals through a maximum of 32 receiving stations has been developed by Schober *et al*. (1983, 2001). It uses the Doppler bearing principle and achieves a maximum bearing accuracy of <1°. The system is compatible with standard VHF tags and can also decode sensor parameters which are encoded through PIM and PDM procedures (Bögel, 1996, 1999). However, during several studies on birds and mammals, a number of drawbacks became also evident: data quality was highly varying and influenced by habitat features (refraction, reflection effects causing multipath wave propagation); furthermore, data plausibilisation was critical due to the enormous amounts of data (up to one data set every 5 seconds) and the highly varying location quality; finally, costs for system installation were extremely high (approx. 20,000€ per station), a fact which can easily exceed the scope of even huge budgets when considering that the practical range of one station is limited to approx. 10 km and thus a network of receiving stations is required to cover a given research area.

Due to their high demands concerning the power output of the transmitters and their implications on tag weight and lifespan, TOA systems never achieved an adequate acceptance level in wildlife tracking applications.

ARGOS Satellite System

The ARGOS system has originally been designed for floating buoys in the ocean and was later on adapted for wildlife applications. The transmitter designs (PTTs) have been dramatically reduced in weight and size so that 20 g PTTs are available. Due to its global coverage, its sufficient lifespan and the acceptable tag weight the system has meanwhile been widely used in many raptor migration studies. The present knowledge on raptor migration would not have been possible without the ARGOS satellite system. However, data processing is expensive, the quality of the locations is often not very satisfying (errors of > 10 km can easily occur; for details see Vincent *et al*. 2002) and the reliability of the units is not always satisfying (especially over Europe due to the increased electromagnetic noise). Although announced quite a while ago, according to the information available to the author, a bi-directional service for wildlife applications is still not supported by ARGOS service, something which would greatly increase energy efficiency and thus lifespan and/or power output could be increased.

GPS-Based Techniques

The main benefit of applying GPS technology to wildlife tracking is the superior data quality with a location accuracy of approx. 10 m in > 90% of the data. In contrast to all other techniques discussed above, the use of this technology for wildlife applications requires a receiver on the animal. Such electronic circuits have become small and light enough for bird applications only recently (they are presently around 100 g and light enough for large raptors). However, the main problem is how to retrieve the data which are logged on the animal and how to cope with the increased demands in terms of weight, size and lifespan which go along with any kind of data download link. Pure dataloggers ("storage telemetry") have only little impact on these parameters but can only be applied to species or specimens which come to fixed places at regular intervals and where the chances for re-trapping and retrieving the units are high. Such places can be nest sites or feeding places but disturbance at such places may be not tolerable. Some companies have developed a timer- or remote-controlled release mechanism to maximize the chance for retrieving the unit. However, these do not always work reliably, and one will have to approach the individual rather closely in order to trigger the radio remote control (some kilometres). Even if the mechanism functions, one must consider that even an opened harness will not drop off immediately but mostly somewhere during flight. So, in any case, you will have a considerable risk of loosing the data. You also have to consider that data-viewing will be a retrospective procedure. Thus, no way of taking action if the data might have indicated that the monitored animal is in need for help. Therefore, most studies require a real-time data access and thus need a kind of communication link between the GPS unit on the bird and a field or base station. Three communication links are being used for this purpose: a) a VHF or UHF radio data link, b) the mobile phone network (GSM), and c) the ARGOS communication link. While option a) is restricted to a rather small range (<10-20 km), option b) is restricted to areas with mobile phone coverage and will create additional costs for data-transfer and option c) is highly cost intensive (hardware and cost for data-processing), heavy and large (>200 g) and still rather new (few experience on reliability). It is worth noting that options b) and c) represent a 100% automated data-sampling, so the collection of animal locations does not require field staff. However, most units also allow for a manual tracking during fieldwork (integrated VHF beacon).

Drawbacks of GPS-based technology are the high costs (> 1,000 € per unit), the relatively short lifespan (500-1000 positions) and the big weight and size. Fig. 1 gives an example of a monitored Eurasian Griffon Vulture *Gyps fulvus* using a GPS-GSM tag.

Conclusions and Recommendations

It is definitely not useful to define a stereotype approach, which will represent the optimal strategy for any project or study. As already stated in the introduction, there are too many aspects to be considered which will affect the proper methodology. However, some general guidelines and recommendations are possible. I will focus these on the practical needs which can be derived from the main objectives of vulture management in the European and Mediterranean range as commonly discussed within the EGVWG (Slotta-Bachmayr *et al.* 2004) and which was also the main focus of this conference.

If the man-power is available, it may be wise to tag some individuals with conventional tags because one usually learns much about a species behaviour and its habitat characteristics during manual tracking out in the field. This is definitely extremely labour-intensive and demanding, but being on the "bird's trace" live is a valuable experience. It is really different if one just has got some coordinates on a computer screen, overlays it with a habitat map and starts analyzing than being out there, finding a bird perching or roosting or feeding and making behavioural observations. In many cases these may be anecdotic events; however, it is likely that one learns more about a study subject than "the bird was using habitat type x at a specific time of the year and day". Also note, that the database in your PC may not include the specific structures, which may be of importance from the animal's point of view.

Due to the arguments presented and discussed above, automatic ground-based stations are probably not very useful for current needs of vulture monitoring. The costs for system installation and maintenance are high and technically highly educated personnel is required. Such systems may still be helpful when studying local aspects of vulture biology, especially if activity, time-budget or eco-physiologic questions are to be addressed. However, in general, the technical evolution of telemetry techniques has evolved towards other directions like ARGOS and GPS-based tracking, where established networks for data-transfer are used which are operated and maintained by international companies.

If migration and movements between sub-populations is of primary importance and relatively long transmitter lifespans have to be obtained (> 1 year), ARGOS is the adequate tracking technology. A wide range of manufacturers offers such tags (for example Northstar and Microwave) and the technology has also been successfully applied to vulture species (Bahat, Woodly and Hatzofe, pers. comm.). Especially for studies in Europe it is wise to order units which work on the frequency bands with a reduced noise level and to choose an increased power output option for the PTTs (250-500 mW) as the power output of many models has been reduced to a critical limit in order to reduce weight and maximise lifespan. However, many tags fail under conditions with a high noise floor (which is typical for the densely populated areas of Europe) and unit failures of around 30% have been reported (McGrady, pers. comm.). It is important to note that you will not get free replacement units from the manufacturer because normally you can never prove the ultimate cause of failure (e.g. the bird might have also been shot!). Since project budgets are usually fixed, such events will simply reduce your sample size!

Concerning GPS-based technologies, the development is still in progress. Units decrease in weight from generation to generation and more and more manufacturers offer such tags for large birds. Currently up to 1000 positions are possible with a tag weight of approx. 200 g which can be distributed over time according to the study needs. While several manufacturers already provide units with VHF/UHF through a radio link or with download through the mobile phone network (for example Vectronics-Aerospace, Televilt/TVP and Lotek), still very few models are available where the download of GPS data is done through the ARGOS link and thus a global coverage is possible without restrictions. However such "cutting edge techniques" are extremely expensive, still relatively heavy and only very limited experience is available for birds. Own experience with GPS-GSM units (data download through the mobile phone network) on Griffon Vultures but also on mammals (such as Alpine ibex Capra ibex) shows that reliability of such techniques can still vary a lot. There are also hints that synchronisation of the tags to foreign mobile phone networks (international roaming) may cause problems. On the other hand, a comparable tag manufactured by Vectronic-Aerospace which is applied to a specimen of a resident Alpine ibex population in Switzerland works extremely reliably for a period of almost 2 years.

In conclusion, I suggest that an intelligent combination of various telemetry techniques can optimise the output of a vulture monitoring project and simultaneously limit the risk which may be encountered when focusing on a highly sophisticated but probably still not very reliable technique. This is also supported by considering practical facts of species conservation, where normally budgets are limited: the alternative may be to invest in a small number of highly sophisticated tags or in a larger number of conventional tags. Provided that sufficient personal resources for fieldwork are available, you can usually monitor a higher number of tagged birds using conventional tags. Especially when one studies habitat use of a local resident population, one can achieve meaningful results with conventional tracking methods. However, this approach is impracticable when focusing on migration, long distance flights and exchanges between sub-populations. ARGOS and/or GPS-based techniques are the suitable methods to address such questions. While ARGOS PTTs do not support adequate location quality for small-scale habitat use and selection, GPS-based tags deliver suitable position quality also for such questions but still suffer from restrictions in terms of lifespan and/or weight.

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EFFECT OF MONITORING FREQUENCY AND TIMING ON ESTIMATES OF ABUNDANCE AND PRODUCTIVITY OF COLONIAL BLACK VULTURES *Aegypius Monachus* IN CENTRAL SPAIN

Javier de la Puente

SEO/BirdLife (Sociedad Española de Ornitología)
Área de Estudio y Seguimiento de Aves
Melquiades Biencinto 34, E-28053 Madrid, Spain
E-mail: jdelapuente@seo.org

Abstract

Detailed information from nine years of monitoring a large colony of Black Vultures in Spain is used to obtain population estimates, breeding data and correction factors concerning the censuses effort and dates of the visits. From the onset of the breeding season, at the beginning of February, detectability of total pairs and breeding pairs increases rapidly until mid-March and then declines until the end of the breeding season. Detectability of breeding and non-breeding pairs differs. The latter exhibit a period of high detectability in March and early April which then decreases until mid-May, being virtually nil afterwards. Nest use is directly related to breeding activity, as higher detectability coincides with the average laying date for the colony. The frequency of visits affects less the estimation of chick numbers. Hatching and fledging phenology determines chick detectability. The number of visits influences the population size estimate for the colony. Fewer visits result in higher estimates of productivity and breeding success. The results of this study show that four or less visits to the nests are insufficient. For such a species, at least eight census visits are recommended, as they permit a reasonably good population estimate to be obtained. The results of this study should be considered when planning census dates and effort for this species. It is conservatively estimated, that the Spanish population of Black Vultures in 2004 comprised at least 1,710 pairs, instead of the 1,523 pairs that the censuses indicated. Moreover, it is expected that the number of fledglings and the breeding parameters (e.g. breeding success) derived for the Spanish population of Black Vulture are over-estimated due to the low number of visits to the colonies.

Keywords: Monitoring Frequency; Census; Population estimates; Breeding parameters; Black Vulture.

Introduction

The setting up of a census to obtain estimates of population size and reproductive data for a particular species is a fundamental part of the basic information for its conservation as well as for its population trends (Donázar & Fernández, 1990; Birdlife International, 2004). However, carrying out censuses on a large geographical scale is very costly, both in terms of finance and manpower. Therefore it is essential to establish a suitable protocol for a good quality census at an affordable cost (Tellería, 1986).

The Black Vulture *Aegypius monachus* is a threatened species throughout most of its range in Europe (BirdLife International, 2004); however, the monitoring programs for this species vary from region to region and from time to time (Sánchez, 2004; Tewes *et al.* 2004). Generally, the design of the census protocol for this species (number of visits, dates, etc.) has been based on the opinion of experts but without adequate, correctly analysed information to support it.

In this work, detailed information from nine years of monitoring a large colony that contains 5% of the Spanish population (more than 70 pairs in recent years; De la Puente, in press) is presented. This information is used to obtain population estimates, breeding data and correction factors relating the census effort and timing of the visits.

With regard to this, the principal objective of this study is to establish which is the optimal time and frequency for census visits in order to obtain the most accurate estimates of population size and breeding parameters. Therefore, information obtained on phenology and monitoring of the colony has been used. Various census efforts are simulated in order to evaluate the results obtained. Correction factors are obtained that can be used to improve population estimates of other colonies not so intensively monitored. Finally, these results are applied to carry out the first refined estimate of the Spanish population of Black Vultures.

Material and Methods

Study Area

The study was carried out at a Black Vulture colony in the Lozoya Valley (Sierra de Guadarrama, Madrid), which is located at the northern limit of the species distribution in Spain (Del Moral & De la Puente, 2005). The colony is found in hills dominated by Scots Pine *Pinus sylvestris* and the nests are at an altitude between 1248 and 1903 meters above sea level. Annual rainfall is about 900 mm and median annual temperature is 9.9^{o}C, and the annual average of number of days with snow cover is about 40 (De la Puente, in press).

Monitoring of the Colony

The usual survey of the colony between 1997 and 2005 in order to find and monitor the breeding pairs, took place four days per week from the beginning of February until the end of September. Observations of all the potential nesting habitat from various watch points were carried out (Fuller & Mosher, 1981).

Attempts were made to check all the nests twice a week. As a result, the breeding phenology (laying, hatching, failure or fledgling dates) of the pairs and that of the colony as a whole were obtained, in general, with great precision. However, between 1997 and 2003, failed and non-breeding pairs were checked until the end of June only. In order to obtain information concerning the condition of all the occupied nests each season during 2004 and 2005, which differed from previous years, all pairs (breeding and non breeding) were surveyed continually from February to September, a time span representing the period from laying of first egg to fledging of the last chick (Del Moral & De la Puente, 2005). On each visit presence and number of birds in the nest was recorded, as well as whether the birds were standing, incubating or attending young.

All this enabled us to know the exact location of each nest and to be sure that no pair had been left unaccounted for.

Phenology and Breeding Data

In order to calculate the phenology of the colony, information obtained over nine years has been used. The date of incubation onset was determined as the median day between the last visit with an empty nest or a standing adult and the first visit with an incubating bird. The hatching date was determined as the average between the last visit with an incubating adult and the first visit with adult and chick. The fledging date was determined as the average day between the last visit to the nest with young present and the first visit to the nest when it was empty. The phenology of failures was taken from De la Puente (in press). The percentage of clutches, hatching and fledged young was calculated weekly from the total number of clutches, hatching and fledged young respectively. With all the dates of laying, hatching and fledging the average values were calculated.

An active breeding pair was defined as one that laid an egg and a non-breeding pair was one that occupied the nest on at least three visits but didn't lay an egg. Breeding success was calculated as the number of fledglings divided by the number of breeding pairs, and productivity was defined as the number of fledglings divided by the total number of pairs (breeding and non-breeding). On each visit it was considered that the nest was occupied by a pair when two adult vultures were at the nest, one was standing and one incubating; one incubating adult was present; one adult with chick or a young chick on its own was present.

Detectability of Pairs and Effect of Census Effort

The number of visits to the nest affects the detectability of the adult pair and its offspring. Because the study was carried out at a single colony, other factors that can influence the results of a census (such as the ability and the experience of the observer, the observer's knowledge of the colony, the type of vegetation where the nests are to be found, etc; Fuller & Mosher, 1981) have not been taken into account.

Out of the total number of pairs found during the intensely monitored years of 2004 and 2005, those that had at least one contact in all the fortnights from the start of February to the end of September were selected. In the case of more than one visit per fortnight, the visit nearest to the 8th or 23rd days of the month were always used. For estimating whether a pair was breeding or non-breeding all the contacts were utilized.

The detectability on a given fortnight was calculated as the percentage of pairs or young detected in relation to the actual number of pairs or young. The real number is that established after the accumulated observations of the maximum number of visits considered (sixteen).

To establish the census effort the following possibilities were considered:

- Two visits: one during incubation and one with young. The fortnights with greatest detectability for pairs and chicks (the second in March and the second in June respectively, Fig. 1) were used.
- Four visits: two during incubation (first in March and first in April) and two with young (second in May and second in July). The dates are selected bearing in mind that during the censuses the checks are usually spread through time and we have tried to choose the periods with most incubating pairs or most young in the nest (Fig. 1 and 2).
- Eight visits: one per month, three or four during incubation and four or five with young depending on the phenology of each pair. The first fortnight for all months has been chosen randomly.
- Sixteen visits: one per fortnight. This represents the maximum number of visits.

These possibilities are considered because they demonstrate well the different options for monitoring. In Spain the pairs at the colonies are usually visited twice (normally one during incubation and another with medium to large sized chicks; Table 1). However, in some of the colonies more effort is made, especially in

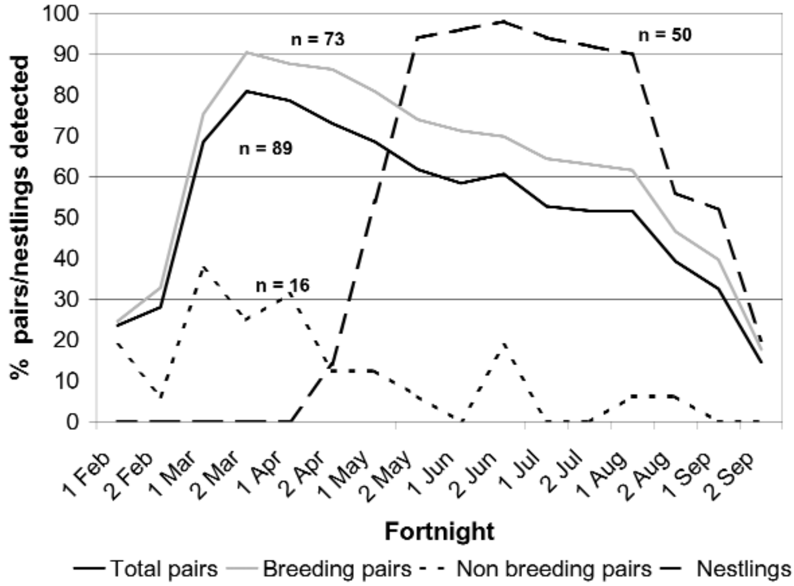

Figure 1. Detectability of pairs and nestlings of Black Vulture between February and September, shown as the percentage of the total sample.

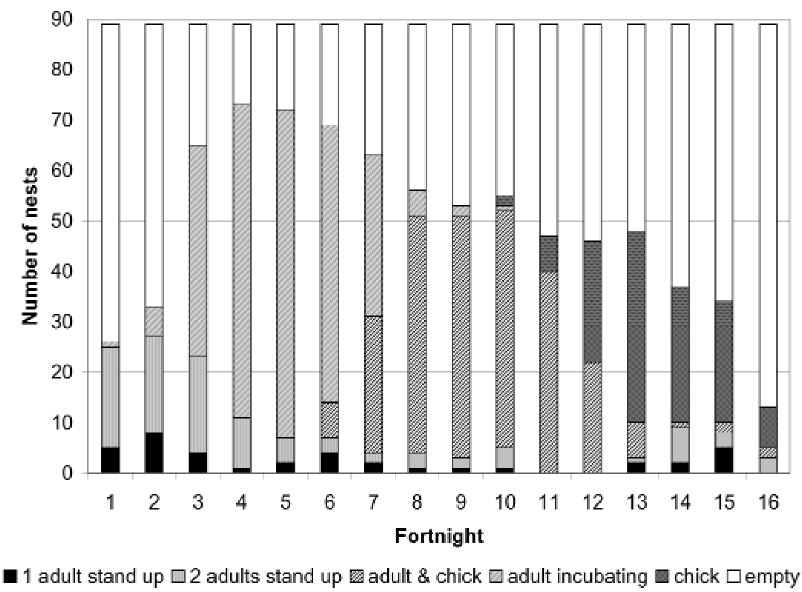

Figure 2. The nest contents for each fortnight throughout the breeding season.

Table 1. Number of visits, pairs detected, correction index and corrected pairs number per each autonomous community with Black Vultures.

	No. visits in census	Total pairs detected	% Pairs corrected	Total pairs
Extremadura	2	719	+ 14.6	824
Castilla y León	2	231	+ 14.6	265
Castilla-La Mancha	2	273	+ 14.6	313
Andalucía	8	209	+ 3.4	216
Madrid	16	81	+ 0.0	81
Baleares	3	10	+ 10.4	11
Total pairs		**1,523**		**1,710**

Andalusia and Madrid (Galán *et al.* 2003; De la Puente, in press). In this way we can compare what normally happens with what would happen with increased survey effort. That said, more than 16 visits are not possible given the resources usually available.

The number of pairs, young and breeding parameters were calculated for each census effort, using the information obtained from each census effort independently (number of breeding pairs, non-breeding pairs and number of young).

Correction of the Spanish Population of the Black Vulture

The information concerning the number of pairs in each autonomous community was taken from Del Moral & De la Puente (2005) and from the conclusions of the International Symposium on the Black Vulture *Aegypius monachus* (De la Puente, 2005). The census effort in each community was facilitated by the different regional governments and Black Vulture surveyors. Sometimes it was necessary to calculate an average effort for the whole autonomous community.

Results

Phenology of the Colony

The average laying date was 10th March (average ± S.D.= 10th March ± 14.4: n = 316, Fig. 3), the earliest date was February 6th and the latest, May 7th. Regarding hatching and fledgling dates, the averages were May 8th (S.D.= 12.4; n = 258) and August 28th (S.D.= 10.8; n = 267) respectively. Most failures took place between the last week of April and throughout May (Fig. 3), coinciding with the hatching of most clutches (De la Puente, in press).

Detectability of Pairs and Effect of Census Effort

The surveys in the years 2004 and 2005 provided a sample size of 89 pairs (73 breeding and 16 non breeding) and 50 chicks, with at least one visit per fortnight (16 visits to each pair).

Detectability of pairs varied over time (Fig. 1). From the start of the breeding season, at the beginning of February, detectability increases rapidly until mid March and then declines continuously until the end of the breeding season. Nevertheless, detectability of breeding and non-breeding pairs is very

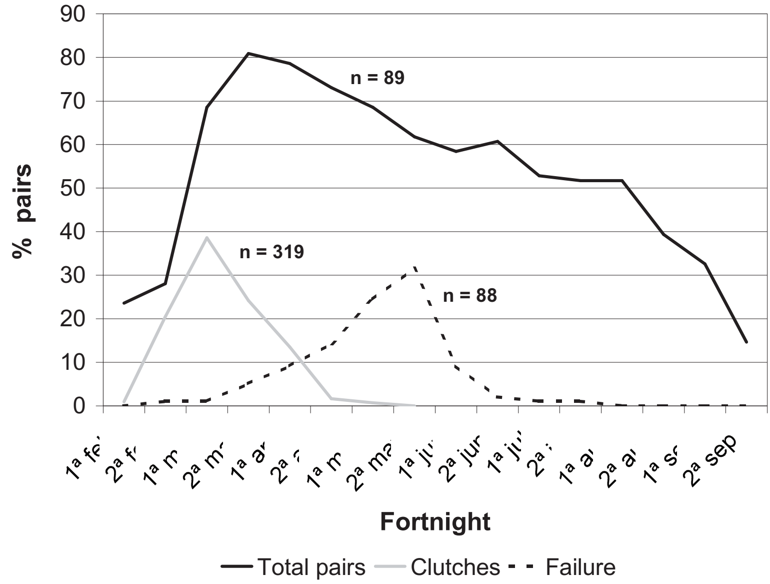

Figure 3. Detectability of pairs, laying date and breeding failures in the study colony. The sample size of clutches and failures has been obtained considering only those with accurate date of clutches or failures between 1997 and 2004 (De la Puente, in press).

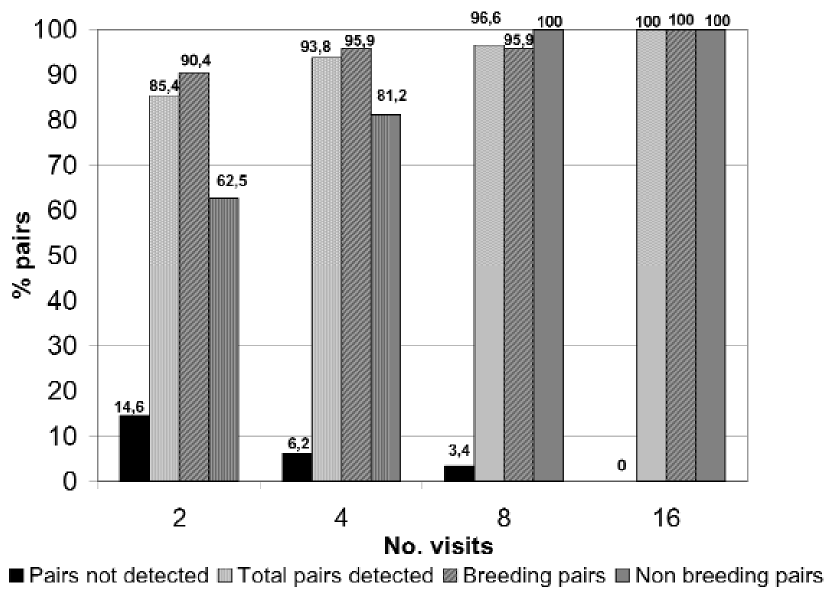

Figure 4. The proportion of pairs detected relative to the number of census visits made.

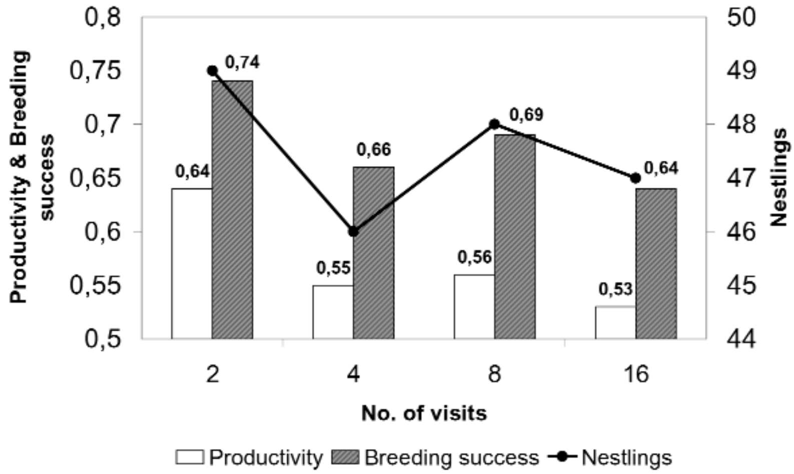

Figure 5. Productivity, breeding success and nestlings detected in relation to number of visits made.

different. The former show the same pattern as the total number of pairs. The latter show a period of highest detectability in March and early April, which then declines until mid May, being virtually zero after that time.

The number of visits influences the population size estimation made for the colony (?23 = 18.023; p<0.001, Fig. 4). With two visits, 14.6% of the pairs are not detected, with four visits 6.2% and with eight visits only 3.4%. Regarding chicks, the frequency of visits is of less importance, however generally, fewer visits result in higher estimates of the number of chicks because losses go undetected. This influences productivity and breeding success estimates (Fig. 5) and indeed, fewer visits result in higher values.

Refining the Estimate of the Spanish Population

This correction factor was calculated from the visits carried out to the colonies in the different autonomous communities (Table 1) and from using the percentages of undetected pairs resulting from minimal census effort. It is estimated that the Spanish population of Black Vulture in 2004 comprised at least 1,710 pairs, instead of the 1,523 pairs estimated from the previous censuses.

Discussion

Nest use is directly related to breeding activity, given that highest detectability coincides with the average laying date for the colony (Fig. 1, 2 and 3). From this time onwards, breeding pairs use the nest less due to two main reasons: firstly, non-breeding pairs particularly frequent the nest in March and first half of April, but after that they use nests very little (Fig. 1), and secondly, failed pairs use the nest also very little.

Failures at the chick stage are infrequent in this species (loss rate average is two chicks per year over an eight year period from a total number of 290 chicks in the colony during this period; De la Puente, in press). Hatching and fledging phenology determines chick detectability, which increases from the first half of April when the first eggs hatch, until the first half of August when the first chicks fledge (Fig. 1 and 2).

It is important to establish the required census effort in order to obtain the highest quality results because this affects the number of pairs and chicks found as well as the breeding parameters. The results of this study show that two or four visits to the colony are insufficient (Fig. 4 and 5). The number of detected pairs between two and 16 visits and between four and 16 vary significantly (Yate's corrected chi square; ?21 = 11.95; $p < 0.001$; ?21 = 4.31; $p < 0.05$ respectively), but not between eight and 16 (Yate's corrected chi square; ?21 = 1.36; $p = 0.244$). For this species, at least eight census visits are recommended, which permit a reasonably good population estimate to be obtained (Fig. 4). During the chick stage the number of visits has a smaller effect because failures are infrequent in the species (De la Puente, in press). Nevertheless, the breeding parameters obtained show a closer fit to the real figures using data from four visits rather than with eight. This is because although the same number of pairs are detected from four and eight visits, with four visits almost 20% of the non-breeders are undetected compared to the results from eight visits (Fig. 4). Therefore, by increasing the number of visits, better population estimates and data for breeding parameters are obtained, increasing the former and decreasing the latter (Fig. 4 and 5). Moreover, with four visits the number of chicks is slightly underestimated and with eight visits it is slightly overestimated (Fig. 5). These are the results that produce the differences. In any case, the differences in the breeding parameters resulting from four and eight visits are very small.

If the colony is to be surveyed fewer times, it is very important to adjust the timing of the visits to the period of highest detectability of the pairs in the nests (Fig. 1 and 2).

The visits should be spread through as long a time period as possible as there can be a different breeding period span of up to two months within the same colony (Hiraldo, 1983, own data). In particular, to reach maximum pair detection it is advisable to try to put extra effort into visiting the colony between March 16th and April 30th, the period of highest pair detectability varying between 73 and 80.9%, and particularly in the second half of March. For the breeding pairs (with clutches) the best time is between March 16th and May 15th, when pair detectability is highest varying between 80.8% and 90.4%, and particularly in the second half of March also. Non-breeding pairs are not easily detected, the best time to do so is the first half of March when still a mere 37.5% are detected, with four visits almost 20% going undetected (Fig. 4). This last group is more prone to underestimation than any other as a result of a reduced number of visits to the colony. The best period to find chicks is between May 16th and July 15th, when chick detectability is highest, varying between 92 and 98%. The best time in particular is the second half of June when up to 98% of the chicks in one season may be found (Fig. 1 and 2).

These results are clearly determined by the average laying date of this colony, in this case being March 10th (Fig. 3); which also determines the hatching and fledging dates. In more southern colonies, particularly Sierra Morena and Salamanca, the phenology can be slightly earlier (Hiraldo, 1983). There may be other phenological differences in other colonies across the distribution area in Europe and Asia. Therefore, it is possible that the census dates should be one or two weeks earlier if there is confirmed knowledge of an earlier laying date. Unfortunately, this kind of information is not available for most colonies (Cramp & Simons, 1980; Del Moral & De la Puente, 2005).

The correction to the Spanish population of Black Vulture has been made in a rather conservative way. In little known colonies or those which exhibit very different parameters from the Madrid colony, the true number of pairs may be strongly underestimated. Moreover, it is expected that the number of fledglings and the breeding parameters derived for the Spanish population (see review in Del Moral & De la Puente, 2005) of Black Vulture are over-estimated due to the low number of visits to the colonies. Thus, for instance, it has been established that in the Cabañeros colony (Ciudad Real) there are 141 pairs of Black Vulture (Del Moral & De la Puente, 2005). However, this figure refers to pairs with fledglings (J. Jiménez, pers. com.). Additionally, it is estimated that 30% of the pairs fail and that there are 12% of pairs that do not breed. Using this information, the number of pairs in Cabañeros could be increased to 243 pairs. The date of the census was not taken into account, even though this would affect the results.

The results of this study should be born in mind when planning visits, dates and effort of censuses for this species. Also it would be very useful to have similar information concerning other colonies under different environmental pressures that might originate from different phenology and detectability. Thus, this would provide some of the required information for carrying out effective censuses correctly when there are insufficient other sources of information available.

Acknowledgements

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SUPPLEMENTARY FEEDING PROGRAMMES: HOW NECESSARY ARE THEY FOR THE MAINTENANCE OF NUMEROUS AND HEALTHY VULTURE POPULATIONS?

Steven E. Piper

School of Biological and Conservation Sciences
University of KwaZulu-Natal, Private Bag X01 Scottsville Pietermaritzburg
3209 KwaZulu-Natal, South Africa, Tel./Fax +27 (0)33 7011 741
E-mail: vulture@telkomsa.net

Abstract

Three forms of supplementary feeding schemes have been used in vulture conservation: pure supplementary feeding, predator simulation and vulture restaurants. The many hypothesized benefits and disadvantages of these are presented in some detail. This is followed by a number of case studies where the provision of supplementary food is claimed to have assisted in population augmentation of vultures. It is concluded that, as yet, the effects and efficacy of supplementary feeding for vulture conservation have not been properly studied.

Keywords: Food supply; Predator simulation; Supplementary feeding; Vulture conservation; Vulture restaurants.

Introduction

There are many examples in the conservation literature where the population decline of a bird species, including vultures, has been attributed to a reduction in the food supply (e.g. Bearded Vulture *Gypaetus barbatus* in the former Cape Province of southern Africa (Brown, 1991) and California Condor *Gymnogyps californus* in North America (Snyder and Snyder, 2005: 61, 96, 122ff, 132 and references therein). As a consequence, the provision of extra or supplementary food has been advocated, e.g. Cape Griffon Vulture *Gyps coprotheres* (Butchart, 1988) and California Condor (Snyder and Snyder, 2005: 61, 96, 122ff, 132 and references therein). In a perfect world, such supplementary food would be made available in the environment in a way that closely matched the spatial and temporal abundance of natural food (Snyder and Snyder, 2005; Anderson and Anthony, 2005).

It is suggested that there are three paradigms for supplementary feeding: *pure supplementary feeding, predator simulation* and *vulture restaurants*. In a *pure supplementary feeding* scheme, food is placed at strategic points in the environment at relatively high frequency. The food supplied is always in addition to the food already available in the environment. Ideally, the spatial and temporal location should both be random so as to discourage habituation. The quantity of food provided should be at least equal to the estimated shortfall and the maximum inter-provision interval should not permit any individuals to starve. The food should be placed at locations that will maximally benefit the target species. This is typically what happens in a formal re-introduction project, e.g. California Condor (Snyder and Snyder, 2005).

In *predator simulation*, herbivores are killed and left where they were found and so are available to all scavengers. While *predator simulation* may aid in vulture conservation, this is usually not its main function (I. Rushworth, pers. comm.). The food provided is always in addition to that already available in the environment, it is provided randomly in space but generally regularly in time (e.g. once a week, month etc.), often at intervals much greater than the time in which a vulture could starve to death. i.e. the vultures could not become dependent on the food source.

The provision of supplementary food for vultures, often at fixed feeding sites, called *vulture restaurants* has been advocated and practised for almost forty years in Africa, Europe, Asia and North America (D.C. Houston, this volume). The food provided at a vulture restaurant may have been brought to the restaurant from where it died and so may not be in addition to what is available naturally. The spatial location of the vulture restaurant is fixed, though the provision could be random in time. The time interval between the arrival of successive carcasses could, on occasions, be much longer than the period in which a vulture could starve; again, this will help to reduce the possibility that vultures could become dependent.

This paper reviews the practice of supplementary feeding for vultures and attempts an evaluation of its advantages, problems and successes.

Methods and Materials

From 2000 to 2003 a systematic and exhaustive survey was undertaken of all known vulture restaurants in southern Africa (Piper, 2004). Subsequently the survey was continued opportunistically (pers. obs.). In parallel with this, an on-going literature review was set in motion and a total of more then 400 references were collected. The review presented here is based on this survey and review and also on personal discussions with operators of supplementary feeding schemes for a number of different vulture species on three continents.

Results

This analysis is divided into three sections: the putative advantages of supplementary feeding schemes, their possible disadvantages and their inferred effects on vulture populations and their conservation.

Advantages

Supplementary feeding schemes have been advocated on the basis of six possible advantages, which are outlined below.

1. The provision of extra or supplementary food. For *pure supplementary* feeding schemes and for *predator simulation* the food supplied is always in addition to what is naturally available. However, there are situations where the establishment of a *vulture restaurant* results in food being made available to vultures when otherwise it would not. It is important to note that this is extra food and is in addition to what they could find on their own. There are three situations in which this could occur. A) At larger abattoirs there are processing plants that take nearly all waste materials and convert it into bone-meal, animal foods or supplements for domestic livestock. However, at small to medium abattoirs this is not done because there is not sufficient throughput to warrant such plants and the disposal of waste to vultures, and other scavengers, is a viable option (G. Bradford, pers. comm.). Near the Waterberg in Namibia there is a large vulture restaurant that often makes available more than 400 kg of offal and other waste meat about once a week, which it gets from a nearby abattoir. This vulture restaurant attracts many hundreds of vultures and other scavengers and has been running successfully for a number of years (Diekmann *et al.* 2004). B) In regions where there are high standards of veterinary and human medicine, carcasses of domestic animals intended for human consumption are occasionally condemned as being unfit for human consumption and the vultures may be used to dispose of this meat. In the Free State Province of South Africa there is a large vulture restaurant along the edge of Sterkfontein Dam that receives all the condemned carcasses from a nearby medium-sized abattoir. This vulture restaurant has been running for almost a decade and attracts many avian scavengers (A. Botha, pers. comm.; pers. obs.). C) In southern Africa there is sometimes a deep divide between landowners and farmers on the one hand and their labourers on the other. The labourers are usually poor and cannot afford to purchase safe meat products and so there is often a conflict over livestock that die from disease or accident. In these situations, a farmer may take a dead animal and dump it at a vulture restaurant rather than give the carcass to the farm labourers. The farmer will argue "if I give this dead animal to my labourers then they will cause more animals to die". This is a fairly common attitude in southern Africa (M. Neethling, pers. comm.; Piper, 2004).

2. As an attractant so as to provide other dietary supplements, e.g. calcium. In the 1970s it was concluded that the high incidence of skeletal abnormalities in Cape Griffon Vulture *Gyps coprotheres* nestlings was due to a lack of calcium in their diet because their parents could not access suitable bone fragments. The lack of bone fragments was ascribed to the elimination of carnivores (especially Lion *Panthera leo* and Spotted Hyena *Crocuta crocuta*) that chew and break bones at carcasses. In an important experiment it was shown that the provision of bone fragments was correlated with a decline in the incidence of skeletal abnormalities in nestlings (Richardson *et al.* 1986). Hence, it has been recommended that bone fragments be provided at all vulture restaurants within the range of the Cape Griffon Vulture (Mundy *et al.* 1992). It is believed that California Condors once took the shells of marine molluscs and bone fragments of small mammals in order to acquire sufficient calcium (Snyder and Snyder, 2005: 48, 167) and so in recent times supplementary feeding for this species is often done using the carcasses of young cattle (Snyder and Snyder, 2005: 43; pers. obs.).

3. To establish a safe place for birds to feed. There are occasions when an animal dies in a place that is dangerous for vultures and it is advisable to move the carcass, if possible, to another place where the scavengers can feed in safety. There are two examples of this, the first is where an animal is killed by a vehicle and is left at the roadside. It is known that in these situations passing vehicles can then kill vultures that come to feed on the carcass (Anonymous, 1967; pers. obs.). The second is where vultures come down to a carcass in full view of passers-by who then attempt to kill one or more vultures to use them for traditional medicine (I. Rushworth, pers. comm.).

4. For the provision of poison-free food and to keep vultures from going into areas where the food is poisoned. From the 1940s through to the 1980s most efforts to conserve the California Condor were directed at the preservation of its habitat (Snyder and Snyder, 2005: 131ff and references therein). However, when it was realised that the population was on the brink of extinction which, it is argued, was caused largely by lead poisoning, a different strategy was called for (Snyder and Snyder, 2005: 212). To prevent California Condors from foraging from carcasses containing lead pellets or bullets vulture restaurants were provisioned with lead-free carcasses. This was done in the 1970s and 1980s before the last free-flying California Condors were brought into captivity and in the 1990s after zoo-bred birds were introduced into the wild. This practice still continues (Snyder and Snyder, 2005). The dramatic discovery that the startling decline of *Gyps* spp. vultures in India was due to a single veterinarian drug, sodium diclofenac (Oaks *et al.* 2004) has caused vulture conservationists to suggest starting vulture restaurants there where only safe food is provisioned (Green *et al.* 2004).

5. To raise awareness among landowners, farmers and the general public. Vulture restaurants are the ideal locality to show vultures to people who have had no previous exposure to avian scavengers. Seeing vultures in close proximity, watching them land and take-off and squabble for food is an awe-inspiring sight and often makes a great impression (A. Botha, pers. comm.; pers. obs.). At the same time, a land-owner, a land-manager or a farmer who has a vulture restaurant on his or her property may develop a sense of pride in 'his, or her vultures' and over time acquires 'ownership' of the vultures (pers. obs.). Once farmers have taken ownership of 'their' vultures they tend to become more active in vulture conservation and tend to change attitudes among their neighbours (H.A. Scott, pers. comm., pers. obs.).

6. As an eco-tourism tool. Hides placed at vulture restaurants are an ideal place from which to view and photograph vultures. The demand for places in hides at vulture restaurants in southern Africa is growing and the oldest vulture restaurant, the so-called 'Lammergeyer hide' at Giant's Castle, attracts many visitors from May to September each year when it is booked most days (S. Krüger, pers. comm.). The fees paid to use the hide bring in a considerable income, as do the accommodation fees paid to the nearby accommodation venues. (Piper & Andersson, in prep.). There are plans afoot to privately build and run at least three vulture restaurants with associated hides along the Drakensberg Mountains for eco-tourism purposes (M. Gemmell, S. Maclean & A. Abrey, pers. comm.).

Difficulties

There are no perfect solutions to ecological problems, in general, and to population conservation problems, in particular. A total of 15 difficulties and/or objections have been raised to the establishment and operation of vulture supplementary feeding schemes. These are expanded upon below.

1. The vultures could become habituated, alternatively the vultures could not become dependent and so wander off and feed elsewhere on poisoned carcasses. There are conservationists who fear that once food is provided regularly at a fixed place then the vultures will become habituated to that place and cease foraging elsewhere (D.C. Houston, this volume). However, it has been observed at a number of sites that experienced, free-flying vultures will continue to forage widely. For example, in Namibia observations of colour-marked and satellite-tracked Cape Griffon, African White-backed and Lappet-faced Vultures has shown that they do not stay at the central restaurant where they were ringed (M.

Diekmann, pers. comm.). This has also been shown for California Condors (Snyder and Snyder, 2005:124). On the other hand, where food is provided at 'hacking' sites for the release of rehabilitated birds or for the introduction of new birds it is found that sooner or later they start to wander off and explore the wider environment (e.g. California Condor Snyder and Snyder, 2005: 217).

2. The initial high enthusiasm to maintain a vulture restaurant as a conservation tool often declines with time. Supplementary feeding schemes set up by people motivated by vulture conservation often fail because the initial enthusiasm dies away. In southern Africa about half of the vulture restaurants initiated eventually cease to be operated on a long-term and sustainable basis (unpublished data). This is especially true if the conservationists are amateurs and if there is no compelling reason, usually economic, to maintain the vulture restaurant. The restaurant is more likely to continue for a long period if it is a 'core conservation function' of a conservation organisation. Those vulture restaurants run by persons or organisations that gain some direct financial benefit are most likely to run for many years (pers. obs.).

3. Maintaining the food-supply can be problematic. If the operator of a supplementary feeding scheme has to purchase the carcasses or has to rely on the donation of carcasses then it becomes difficult to maintain the supply of supplementary food for a long period of time (Scott, 1986).

4. The vulture restaurant can attract problem animals. While a supplementary feeding scheme may be established for one target species it may attract many other species of scavengers which for one or other reason are not favoured (D.C. Houston, this volume). The three most important examples of this in southern Africa are feral dogs, jackals and other mammalian scavengers and corvids. Feral dogs were reported as a pest at 14.2% of vulture restaurants in southern Africa (Piper, 2004). A vulture restaurant situated in a stock farming region may attract jackals that are much feared by small-stock farmers and were considered a pest at 9% of vulture restaurants in southern Africa (Piper, 2004). It is recommended that vulture restaurants, especially those located in small-stock farming areas, be surrounded by a mammal-proof fence (Mundy *et al.* 1992). Eco-tourists, especially photographers to the vulture restaurant at Giant's Castle complain about the high numbers of White-necked Ravens *Corvus albicollis* attending the restaurant, taking the meat put out for the vultures and harassing the vultures, specially the timid Bearded Vultures (Balfour, 2005).

5. The vulture restaurant can be a source of osteophagy. An unfenced vulture restaurant located in a large-stock farming area is a source of bones which some livestock, especially cattle, pick up and eat, a process known as osteophagy. This can result in the livestock choking on the bones and dying (Piper, 2004). A simple solution is to put a livestock proof around the vulture restaurant.

6. Vulture restaurants may be viewed as vectors in the spread of disease. Livestock farmers and veterinarians often express the view that vulture restaurants are ideal place for vultures to pick up, exchange and spread disease organisms. In this regard, in southern Africa the greatest fears are expressed in connection with 'foot and mouth', Rindepest and Anthrax (unpublished data).

7. Vulture restaurants may be established for financial gain rather than for the benefit of the vultures, and this may noted by the animal rights campaigners. Recent proposals to establish 'birding routes' in southern Africa to boost low-cost avian tourism have advocated the establishment of vulture restaurants as an attractant for bird-watchers to easily see vultures and other avian scavengers (D. Pritchard, pers. comm.). However, there is a fear that this may be a case of doing the 'right thing for the wrong reason' and it may attract the attention of the animal rights lobby who could then attempt to force the closure of all vulture restaurants because they fear that vulture restaurants could cause vultures to become dependent on the food supplied (A. Botha, pers. comm.).

8. If vultures adopt a single drinking trough as their favourite/traditional this may anger landowners because they foul the water provided for domestic stock and also vultures can drown therein. In livestock farming areas the establishment of a vulture restaurant too close to a livestock drinking trough may allow the vultures to colonise and use it for drinking, bathing and socialising. When vultures use a single drinking trough frequently they tend to defecate in and around the trough and over time foul it to the extent that livestock will not use it, this was recorded as an issue by 4.5% of vulture restaurant operators in southern Africa (Piper, 2004). The farmer may then act against the vultures (A. Botha, pers. comm.). In addition, vultures and other raptors have been reported to drown in such reservoirs (Anderson *et al.* 1999).

9. If a vulture restaurant is placed too close to power-lines they could attract vultures to perch on them and so lead to electrocution, collision, fouling and streamer-based flash-over. The electrocution of vultures on power-lines in southern Africa was first noted in the early 1970s (Markus, 1972) and was then systematically investigated (Ledger and Annegarn, 1981). The national electricity supplier in South Africa, Eskom, has strongly recommended that NO vulture restaurant be placed within 5 km, or even 10 km of a power-line (C. van Rooyen, pers. comm.). In southern Africa 21% of vulture restaurant operators rated power-lines as a serious problem (Piper, 2004).

10. If carcasses dumped contain poisons from 'humane' euthanasia (e.g. barbiturates) or other chemicals used in a veterinary sense, e.g. sodium diclofenac, or black plastic bags then this could lead to the deaths of vultures. In areas where First World veterinary medicine is practised domestic livestock are often subjected to many different drugs during their lifetimes. When it is not possible to 'save' an ailing beast and when it is not possible to get it to an abattoir to slaughter it, it may be necessary to subject the beast to 'humane euthanasia' and this is often done using barbiturates or other drugs (unpublished data). Alternatively, the beast is shot but the lead bullet is not removed (e.g. California Condor, Snyder and Snyder, 2005). Such carcasses placed at vulture restaurants can cause losses among scavengers. Poisoning, in all its forms, is regarded as the most serious problem facing vultures in southern Africa (Mundy, 2000) and in India (Green *et al.* 2004). The dumping of offal and other unwanted meat products in plastic bags at vulture restaurants is regarded as a problem in southern Africa (Piper, 2004).

11. Too many visitors can cause disturbance. There is an expectation, from some bird-watchers and tour-guides, that vultures can be seen 'on demand', like water-birds at a dam (A. Botha, pers. comm.). This is not so. Vultures are by their very nature cautious creatures sensitive to disturbance and reluctant to come down to a carcass if there is too much noise or disturbance (Mundy *et al.* 1992). Ideally, viewers in a hide should be seated there before the carcass is put out and before the vultures arrive and they should remain there until such time as the vultures have finished eating and have departed. One the other hand, there are vulture restaurants and hides where vultures have been habituated to people, one only needs to think of the many carcass dumps in India and the 'sky-burial' sites in Tibet (Harrer, 1956; pers. obs.); however, those are among strictly non-violent peoples.

12. An over-supply of food can lead to a build up of non-target species. If more food is put out than the vultures can handle then there is a real possibility that the carcasses will decay and attract flies and other non-target species. A build up of invertebrates can attract an additional set of birds including insectivores, e.g. Helmeted Guineafowl *Numida meleagris* (Komen, 1984; A. Botha, pers. comm.).

13. Vulture restaurants are 'not natural'. An objection by some conservationists is that vultures have existed for millions of years by finding their own food and the provision of supplementary food, especially at vulture restaurants is 'not natural'. This argument runs at two levels, the first is an aesthetic one in which only 'pure' or 'natural' solutions are permitted while the second is more fundamental, arguing that if vultures cannot survive on the existing food supply them they should not kept alive in a world in which they no longer have a self-sustaining ecological role. However, it is likely that there are a number of vulture species that are largely dependent on domestic stock throughout much of their current ranges, e.g. Cape Griffon, Eurasian Griffon etc.

14. Fences around vulture restaurants can be a problem. Fences placed in a vulture's flight path have been noted as a cause of mortality (Benson and Dobbs, 1985) and were regarded as a problem by 1.3% of vulture restaurant operators in southern Africa (Piper, 2004).

15. The theft of meat by local inhabitants. In southern Africa this is a serious problem having been recorded at 21% of vulture restaurants (Piper, 2004).

Case Studies

The provision of supplementary food has been advocated for the conservation of a number of vulture populations around the world and in some cases this has been a success.

1. Cape Griffon Vulture in the south-western Cape, South Africa. In 1984 a supplementary feeding scheme was started for the Cape Griffon Vulture deme centred on the Potberg breeding colony where food was put out during the summer, when it was in short supply, and when most deaths of juveniles were recorded. Subsequent to this there was a statistically significant increase in the survival of first-year birds (Piper *et al.* 1999) and the colony has grown steadily ever since (H.A. Scott & K. Shaw, pers. comm.). Further north, in the former Transvaal Province, the establishment of a new vulture restaurant has been claimed to be the reason of an almost abandoned breeding Cape Griffon Vulture colony (i.e. Nooitgedacht on the Magaliesburg Mountains) expanding rapidly for over a decade (Verdoorn, 1997).

2. Vultures in the south of France. One of the tools used in the re-introduction of Eurasian Griffons (*Gyps fulvus*) in the south of France was supplementary feeding and this population has grown organically (i.e. from within) and has also attracted other individuals from elsewhere in southern Europe (Sarrazin, 1998; Terrasse, 1985; Terrasse *et al.* 1994 and Terrasse, this volume).

3. Bearded Vulture in the Alps. From the very beginning of the project, the use of supplementary food at hacking sites was an integral part of the re-introduction programme for Bearded Vultures into the Alps (Frey, 1992) and this programme has been highly successful (Frey *et al.* 2004).

4. Vultures in Israel. Egyptian Vultures and other species have been encouraged to use parts of the Negev Desert, and other places in Israel, as a result of the provision of supplementary food (Archibald, 1977; Meretsky and Mannan, 1999).

5. Vultures in India. As a result of the drastic and dramatic decline of vultures in India and Pakistan the use of vulture restaurants to provide 'clean' (i.e. poison-free) carcasses has been advocated (D. Pain, pers. comm.).

6. Predator simulation at Ithala, South Africa, has lead to vultures breeding. At the Ithala Game Reserve in the north-west corner of the KwaZulu-Natal Province of South Africa, close to Swaziland, the provision of additional large antelope carcases, about one or two a month has been followed by the appearance of more vultures in the area and also by at least one species breeding for the first time (I. Rushworth, pers. comm.).

7. California Condor re-introduction. The provision of supplementary food has been used in three different ways during the campaign to conserve the California Condor (Snyder and Snyder, 2005). A) During the

1980s, before the remaining few birds were brought into captivity, food was put out to try and stem the decline. B) When naïve birds were being introduced into the wild, using hacking cages, they were fed 'safe' food (i.e. free of lead and other poisons and also at sites free of human 'junk', e.g. bottle tops, glass shards etc.). C) When it was suspected that there was an excess of lead-poisoned carcases in the countryside, again clean food was provided for the Condors (Snyder and Snyder, 2005).

8. Other examples. During this conference there have been a number of examples of vulture conservation projects in which supplementary feeding, in one form or another, has been used as a conservation tool, see for example the papers on Italy and the Dadia Forest in Greece.

Discussion

At the outset it must be said that the provision of supplementary food will only contribute to the conservation of a species if food is the crucial limiting factor. If the population is limited by poison then the provision of clean food will only contribute if it can be ensured that the vultures will not consume any poisoned food. In some cases, if the poison is lethal, diffused throughout the carcass and not biodegradable (in either the carcass or the vulture) then there need only be one poisoned carcass in about 250 for the entire population to be extirpated (Green *et al.* 2004). The provision of supplementary food must be accompanied by a well though out action plan that simultaneously deals with the other important population threats.

Supplementary feeding programmes must be implemented with a careful understanding of the demography of the species and its social structure. For instance, the regular provision of small quantities of food at a few fixed sites in the Negev Desert was of greater benefit to adult birds while the provision, randomly in space and time, of a few large carcasses was of much greater benefit to immature and subdominant individuals (Meretsky and Mannan, 1999). In France there has been a move away from a few large vulture restaurants to a number of smaller sites (so-called 'lite'!) where food is put out in small parcels and much less frequently than at the large sites (F. Sarrazin & M. Terrasse, pers. comm.). This has encouraged the vultures to maintain a wide foraging network (*ibid*.)

The long-term effects of supplementary feeding schemes have not been well studied (D.C. Houston, this volume) and there are a number of philosophical issues, especially the 'naturalness' of perpetual feeding (van Rooyen and Vernon, 1997). These issues need to be carefully examined.

There are at least two ways in which the provision of supplementary food may actual impact negatively on a population. Firstly, there are the various factors listed above, e.g. occasional poisoning, the transmission of disease etc. Secondly, there is a school of 'do-gooders', 'vulture-huggers', call them what you will, who feel a deep sense of satisfaction when they put out food for another species. (One has only to look at garden-bird feeders in the First World! Among these people may be found the 'animal rehabilitators.) Unfortunately, the vast majority of these people, and some of them hold senior posts in formal and volunteer conservation organisations, do nothing to address the real conservation issues that vultures face.

Lastly, the proliferation of supplementary feeding schemes around the world, most proposed as the primary conservation tool, places on us the serious imperative to understand better the positive and negative impacts that these schemes are having and may have in the future.

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HABITAT MODELLING FOR THE BLACK VULTURE *Aegypius Monachus* IN THE CABAÑEROS NATIONAL PARK

José Jiménez García-Herrera

Parque Nacional de Cabañeros, Centro Administrativo
13194 Pueblonuevo del Bullaque, Ciudad Real, SPAIN
Tel: 34 926783456, Fax: 34 926783484
E-mail: jjimenez@oapn.mma.es

Abstract

I studied the growth of the Black Vulture population within the Cabaperos National Park (CNP), Spain. The breeding pairs increased from 62 in 1988 to 144 in 2004. Their spatial distribution patterns were analysed with statistical methods based on geographical information systems (GIS) techniques.

I used this technology to prepare a model of the breeding habitat. To do so, I used generalized additive models (GAMs) for establishing relationships between a response variable, i.e. nest location and a number of spatial predictors. Thus, through multiple regressions I established relationships between the presence of nests and a number of environmental variables. I also quantified how much each predictor contributes to explaining the distribution of the response variable. All the data employed for the definition of the spatial variables have been drawn out of the GIS. Following the establishment of the relationship between the response variable and the environmental predictors, we checked whether the predictive capacity of the model was acceptable. The statistical information was translated into the spatial context by employing GIS. I concluded that, according to the model, the entire potential nesting area is already occupied and a future increase of the population inside the CNP would mean an increase in nest density and not an expansion of the nesting areas. I employed statistical modelling as a means of emphasizing the issues pertaining to the conservation of the species and I discuss which are the important predictor variables that can be managed efficiently. The most important result is the negative effect of roads, which reduce the available nesting area.

Keywords: Cabañeros; Habitat modelling; Black Vulture; *Aegypius monachus*.

Introduction

The Black Vulture *Aegypius monachus* is one of the most important bird species in the CNP, where it has the second largest breeding area in Europe and probably in the (Jiménez, 1998). In this study I modelled the growth of the breeding population and I also investigate which are the most suitable conservation measures. I started by examining the development of the Black Vulture population within the CNP and by modelling its breeding habitat. This required employment of spatially explicit models for getting an acceptable level of realism in prediction and a combination of statistical tools with geographical information systems (GIS). The principal difference with other modelling works, is the high-resolution analysis (Seone *et al.* 2006), which can be applied to the management, as I attempted trying in this case.

Materials and Methods

CNP lies in Montes de Toledo and has an area of 40.856 ha. It is located NE from Ciudad Real and SW from Toledo (North: 39° 35' 11"; South: 39° 16' 41"; East: -04° 14' 49"; West: -04° 40' 40"). The altitude ranges from 520 m to 1,448 m a.s.l., with a mean 788 m. In the meso-mediterranean zone (500-1,000 m) it is covered with a mosaic of well-preserved woody patches dominated by *Quercus suber*, while *Quercus ilex* prevails in south facing slopes and *Quercus faginea* in the north facing ones. These patches also include various successive stages, with *Arbutus unedo*, *Erica* sp., and *Cistus* sp. The dominant tree species in the supra-mediterranean zone (1000-1448 m) are *Quercus ilex* and *Quercus pyrenaica* (Jiménez, 1998).

Figure 1. Location of the study area (CNP).

The locations of 939 nests-years during the period 1998-2004, were used to define the nesting area. The variables considered in the analysis were obtained from the digital elevation model (*Altitude, Slope* and *Orientation*) and from the maps generated with the help of the CNP GIS unit, by digitizing ortho-photographs with 0.5 m on-the-ground resolution (*Vegetation*). Also the corresponding spatial analyses have been made to obtain buffers in raster format (DVias = *Distance to forest roads* and DPueblos = *Distance to nearest village*). The GIS include for analyses all those roads outside the Park but close to its boundary

The statistical analyses have been made with the statistical package *R.R* is a free version of a commercial statistical language/package S-Plus, and can be downloaded from CRAN (Comprehensive R Archive Network) at *http://cran.r-project.org/.*

Previous spatial analyses to the model were carried using GIS software ArcGIS-ArcInfo v.9.0 (Environmental Systems Research Institute, USA) tools as Hawth's Analysis Tools (Beyer, 2004) extension and others developed by the author.

The analysis of habitat through the model consists of comparing the availability and the use to determine the preference for or rejection of a particular habitat (White & Garrott, 1990). The selection of a habitat normally leads to an out of proportion use respect to its availability (White & Garrott, 1990; Apps, 1996). This selection occurs just as much at the global level as at the local level of distribution, since within the local distribution there is a selection of certain components of the micro-habitat. Thus, it is obvious that the selection criteria depend on the level of scale. The differential use of the resources will provide us with essential information on the requirements of the birds and on the management measures needed to be employed (Jiménez, 2002), taking into account the precautions that some authors have indicated (Seoane & Bustamante, 2001).

The generalized additive models (GAM) used for their advantages in modelling (Hastie & Tibshirani, 1990; Lehmann *et al.* 2002), use non-parametric model techniques to estimate the non-linear relations between the response-variable and the predicting variables in an additive model (1).

$$g(\mu) = \alpha + \sum_{j=1}^{p} f_j(x_j)$$

$g(\mu)$ as an additive combination of arbitrary functions of the x_j. The estimated functions f_j are the analogues of the coefficients in linear regression, and represent arbitrary trends that can be estimated by lowess or smoothing splines.

To calculate the habitat model I used the GRASP package (Lehmann *et al.* 2002) found in R, which is a method for producing spatial predictions using GAM.

Results

1. Breeding Population Analysis

The breeding population size (1988-2005) was examined first because of important fluctuations and a recent apparent slowing down in the growth rate (Fig. 2) is observed. For doing so I have used several tests and statistical descriptors (Morris & Doak, 2002).

1a. Population Growth

The study of the population's growth is based on growth rates, times series and the average rate of change is itself changing over time. The growth rates studied are the mean growth ratio μ, and the variance in the growth ratio σ^2.

Figure 2. Annual variation in numbers of fledglings of *Aegypius monachus* in the Cabañeros National Park (1988-2005).

If μ and σ^2 are significantly greater than zero then I can generally argue that the population is growing. It is possible to estimate μ as the mean of r_t and σ^2 as its variance; so, μ:

$$\mu = \frac{\sum_{t=1}^{T} r_t}{T}$$

Where: $r_t = \log \lambda_t$, and: $\lambda_t = N_t / N_{t+1}$, where is the population of year t and N_{t+1} is the population of year $t+1$. Similarly, the variance of r_i, σ^2, may be computed.

From what is already mentioned, I got a value of 0.044 for μ and 0.029 for σ^2. Both are positive, but very close to zero. The 95% confidence interval using χ^2 is: $0.028<\mu<0.060$ and $0.020<\sigma^2<0.049$.

I subsequently studied the time series (1988-2005) by Box-Jenkins ARIMA model (Auto Regressive Integrated Moving Average). This model makes a prediction based on what has already occurred in the past. For that it considers observed values in previous periods (autoregressive component) and the random factors, or the innovation of each (mobile mean component). The last one makes reference, for each period, to the difference between the observed value in that same period and the predicted value, based on the data of previous years (Fig. 3). The predictive power of the model is poor (confidence limits are wide) but it shows a stagnation of the population in the near future.

From the Analysis of Autocorrelations (ACF) I cannot find a significant autocorrelation (Fig. 4). I conclude there is no periodicity in the breeding time series.

I studied μ tendency (growth rate) by establishing a regression of r over time. Though μ decreases, decrease is not significant (R = -0.359). I studied σ^2 tendency (variance of growing ratio development) by making a regression of the quadratic remainders r over

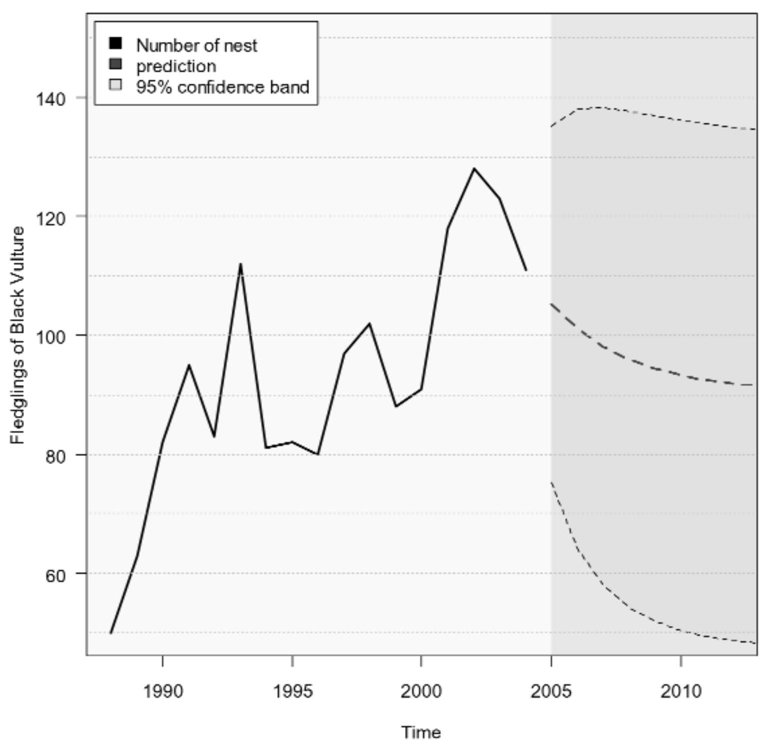

Figure 3. Prediction model for the Black Vulture in CNP.

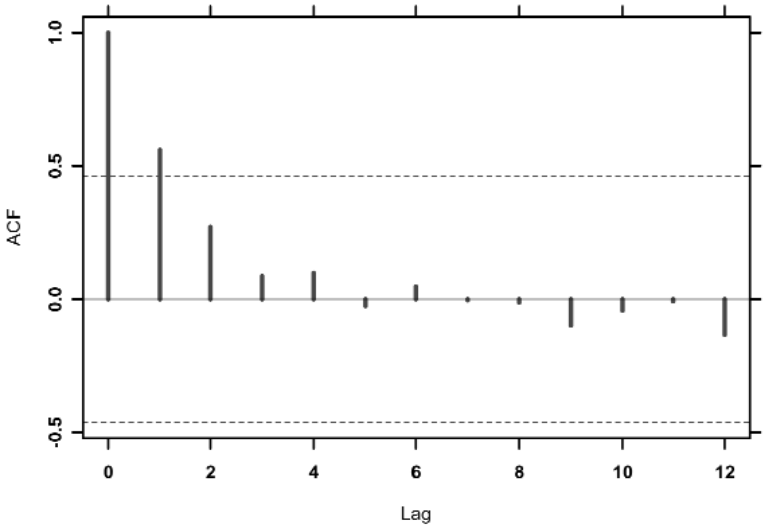

Figure 4. Auto-Correlation Function. It is the correlation coefficients considered as a whole from 1 to half of the series values. No periodic fluctuations of the population are observed.

time. In accordance with the previous result, for σ^2 I see a similar declining trend, but decrease was not significant (R = -0.297).

1b. Density-Dependence Test

I have used a test that consists of fitting a regression of r_i versus N (Fig. 5) ($F_{1,15}$ = 10.14; p = 0.0062). I could argue for the existence of density-dependence Eberhardt (1970) has set at 0.7 the threshold in order to establish a density-dependent relationship. In out case, R = -0.635 (Fig. 5) is quite close to the number given. This result is coherent with the tendencies of μ and σ^2.

Density-dependence can express itself through competition or predation (Hixon *et al.* 2002). In our case, predation is occasional (Del Moral & De la Puente, 2005), so an explanation must be sought in competition. Competition occurs when there is a lack of resources. The availability of food and of nest sites has been documented to be the leading factors to explain density-dependence phenomena (Newton, 1991; Thirgood, 2000; Wiklund, 2001; Krüger, 2002). The possibility of food availability being the carrying capacity determining factor is not consistent. Data from Guzmán & Jiménez (1998) show how food seems more than enough. Therefore I have studied only the availability of breeding habitat.

2. Habitat Model

A 100 100 m grid was superimposed on the area (39,976 cells). The sample size within the Park is 10.000 cells. The centroid of each cell was selected as a point. So each cell takes one value for each independent variable and the response value is “1” (nest presence) or “0” (nest absence). Both, the value of the independent and the response variables have been extracted from the GIS.

The predictor variables used were: *Elevation, Slope, Aspect, Vegetation (vegetCL), DVias* (distance to forest tracks) and *DPueblos* (distance to nearest village).

2a. Data Exploration

I described the environmental space occupied by *A. monachus* with histograms (Fig. 6). In these it can be seen that the nesting habitat of

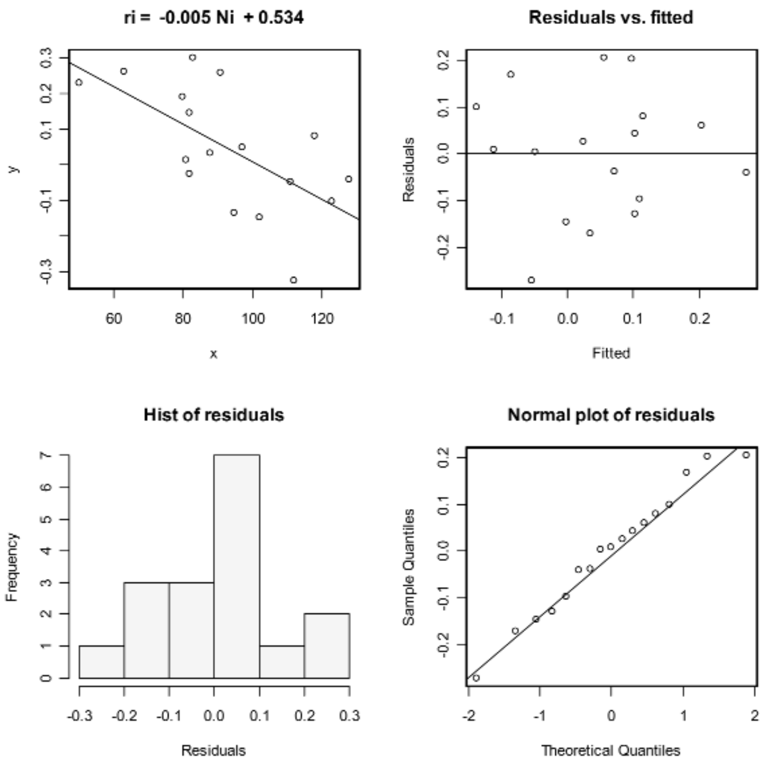

Figure 5. ANOVA test assumptions over the regression. The upper left is a regression plot. The upper right is a plot of residuals versus the fitted values. The residuals are scattered about the line y= 0 with normally distributed values. This is confirmed with the lower left plot (histogram of the residuals). This appears bell shaped. In the normal plot of residuals (lower right), no outliers are detected.

A. monachus is characterized by the absence of forest tracks, a medium to high altitude, mainly southern exposures, medium to high slopes, quite a distance far from villages and never in grassland.

The classes preferred in vegetation are as follows: the most used are Class 1 (dense forest); Class 4 (high bushes); Class 5 (stony areas); Class 9 (sparse forest) and for Class 8 (bushes with trees) is less representative and always over *Q. suber* (76.04%) and *Q. ilex* (23.95%), according to Jiménez (1990).

Correlations between the chosen independent variables were calculated to allow removal of non-independent predictors. This was useful because correlated independent variables can cause trouble in estimating additive surfaces (Lehmann *et al.* 2004). The highest correlation (r^2=0.35) was observed between *Altitude* and *Slope*. However, this correlation was not high enough to justify removing one of these variables from the modelling process.

2b. Model Selection

The stepwise selection with GRASP, that used an approximation of Akaike's Information Criterion (Akaike, 1978) of statistically significant predictors for *A. monachus* breeding distribution, selected the following model (deviance explained = 40.7%):

Sp1: s(DVias,4)+s(Altitude,4)+s(Aspect,4)+ s(DPueblos,4)+s(vegelCL,4)

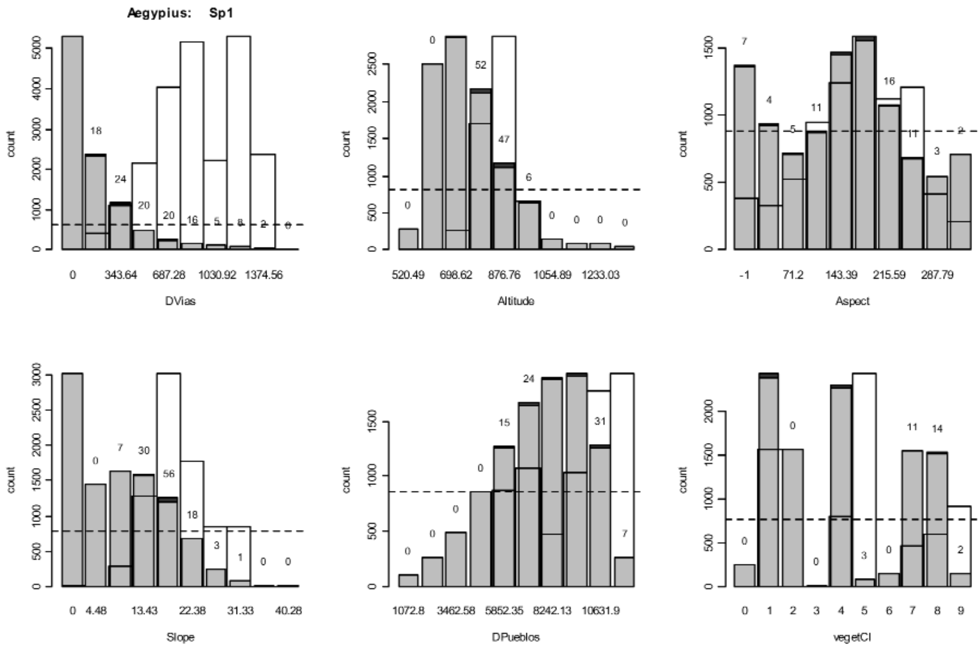

Figure 6. Histograms of predictor variables included in model. The gray portion of the bar shows the cells without nest, and the black portion shows the cells with nest. The solid line shows the proportion of the cell in each bar that have one or more nests relative to the overall average proportion of cells with nests, which is depicted by the horizontal dashed line. Thus, when the solid line is above the dashed line, nests are more common in that portion of the environmental variable. In *vegetCl* the use of class number 5 is relevant, corresponding to isolated trees in stony areas.

Sp_1 represents presence, $s(X,n_i)$ or absence, are smooth functions of the X predictor variables and n_i is the degrees of freedom for the spline smoother.

2c. Model Validation

To assess the accuracy of each model, I estimated the area under the curve (AUC) of Relative Operating Characteristic (ROC) plots (Swets, 1988; Murtaugh, 1996). Validation (ROC=0.952) and cross validation (cvROC=0.94) showed high values and the difference between COR (Spearman Correlation) and cvCOR was small (Fig. 7), confirming the stability of the model (Fielding & Bell, 1997).

4. Model Interpretation

The model displays a prevalence of 116 positive answer variables among 10,000 sampled polygons (1.16% of the sampling polygons have nest).

The predictor's partial response curves are really interesting. *A. monachus* is characterized by a strong positive response to *DVias* (Fig. 8) and by a positive response to medium slope (15-35°), to medium altitude (700 -950 m), to slopes between 15 and 25 degrees, and to *DPueblos* (> 6000). It also has a positive response to south orientation.

The inspection of the contribution of each predictor to the A. monachus model confirmed the dominant role of DVias (Table 3).

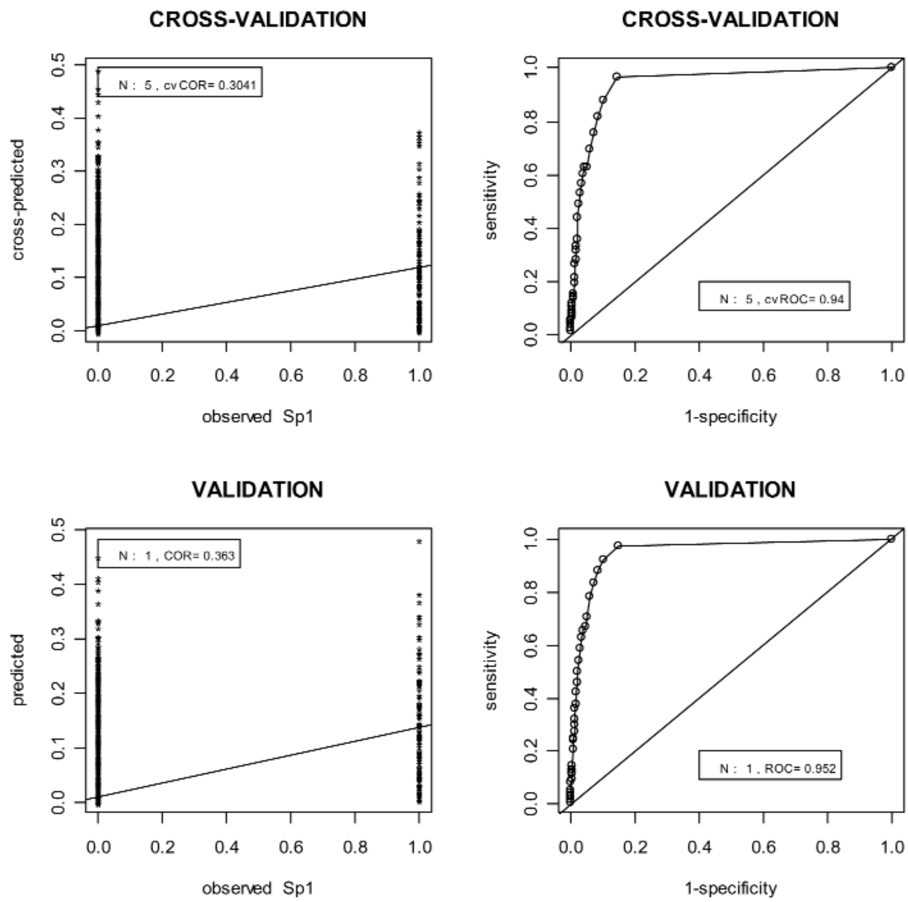

Figure 7. Validation of model. AUC (Area Under Curve) values for ROC and cross-validated ROC (cvROC). ROC>0.9, and consequently the model shows very good discrimination capacity. The significantly higher value of cvROC denotes high model stability.

Table 1. Contributions of predictors to the model.

Predictors	Contribution
DVias	88.30
Slope	56.71
Aspect	53.60
Altitude	30.10
DPueblos	29.46
vegetCl	23.90

have been superposed and analysed in the GIS, and of the result it is possible to infer that practically all the habitat available is occupied.

Discussion

The population analysis sustains the hypothesis of density-dependence of a competitive character in recent growth of the population, where the carrying capacity could come determined by the availability from places adapted

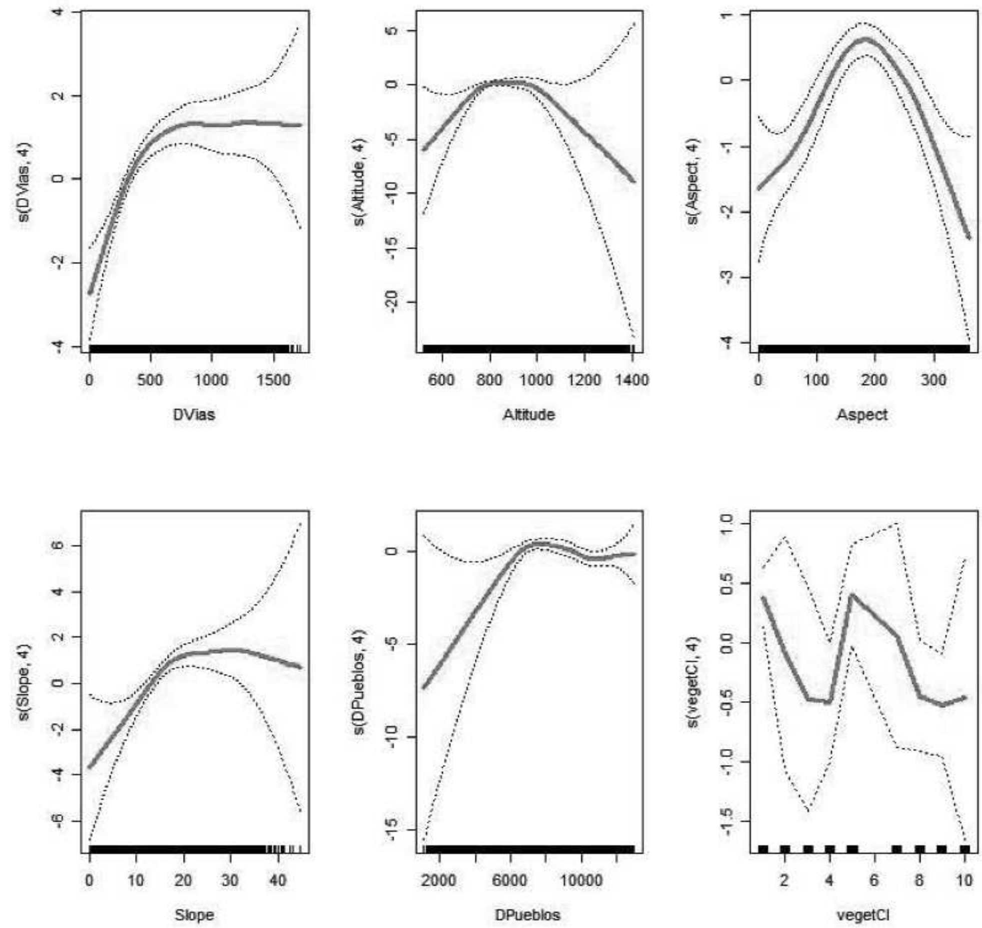

Figure 8. Partial response curves of the presence/absence to the predictors variables in the GAM analysis (center line). The y-axes are based in partial residuals, and indicate the relative influence of each explanatory variable on the prediction. The important contribution of DVias to the model can be seen.

for breeding. Once I have created the model to evaluate the breeding habitat availability in CNP, several conclusions may be drawn.

1. What I first see in the model is that there is a strong positive relationship between distance to forest roads (*DVias*) and nest presence. Other important predictors in the models are *Slope* and *Aspect*. The other variables initially considered in the model (*Altitude, DPueblos* and *VegetCl*) were proved to be of a lower weight.
2. According to the model, the unoccupied optimum nesting habitat for the Black Vulture inside CNP is minimal (Fig.10). The areas evaluated to be as of the optimum quality, host the majority of the nests. The optimal nesting habitats in CNP supports an average density of 2.23 nests/100 ha and the sub-optimal ones host 0.17 nests/100 ha.
3. So, it seems that an increase in the number of fledged young inside the CNP should be

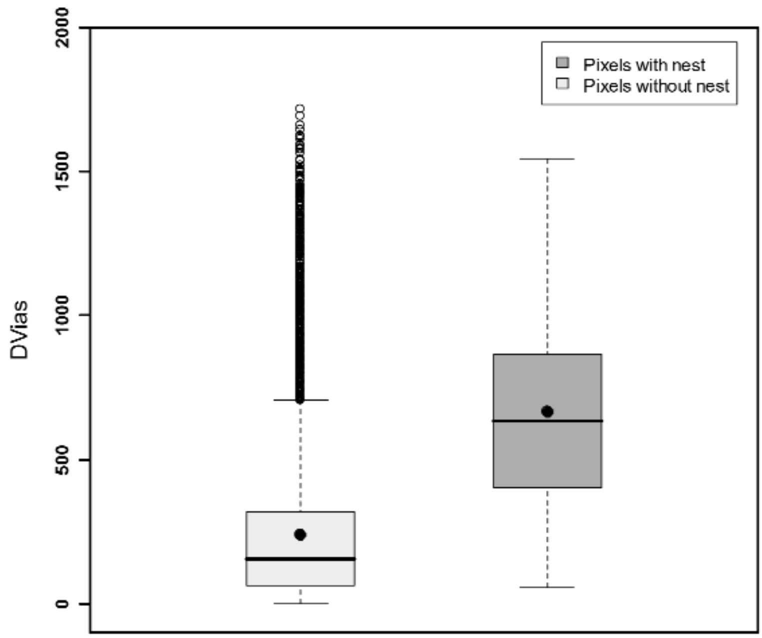

Figure 9. Box Plot of the *Sp1* (response variable) and the predictor *DVias* (distance to forest tracks). Mean distance from cells with nest to tracks is 665.4 m. Mean distance from cells without nest to tracks is 238.9 m.

Figure 10. Suitability habitat for Black Vulture in CNP according to the model developed in this study. The most suitable the nestling areas the darker they appear.

connected to an increase of nest density, rather than an expansion of the existing colony. Density of forest roads seems to be the factor that can be easily kept low as proximity to forest roads apparently reduces the potential nesting habitat for Black Vultures.

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HABITAT ASSESSMENT FOR THE REINTRODUCTION OF THE BEARDED VULTURE *Gypaetus barbatus* IN ANDALUSIA (SOUTH SPAIN)

José M. Padial[1], Mariló Romero-Pujante[2], Francisco J. Hernández[3], José A. Donázar[4], Alfonso Godino[3], Antonio Carrasco[1], Sergio Couto[1], José E. Gutiérrez[1], Antonio Ruiz[3], María J. García-Baquero[1] & Miguel A. Simón[1,3*]

[1] Fundación Gypaetus, C/ Cataluña 8, 1A 23009, Jaén, Spain
[2] Konrad Lorenz Institute for Ethology, Austrian Academy of Sciences
Savoyenstrasse 1a, A-1160, Vienna, Austria
[3] Consejería de Medio Ambiente, Junta de Andalucía,
C/Fte del Serbo 3, 23071 Jaén, Spain
[4] Estación Biológica de Doñana-CSIC, Avda. María Luisa s/n,
Pabellón del Perú, 41013 Sevilla, Spain
[*] Author for correspondence / E-mail: msimon@gypaetus.org

Abstract

In order to assess the availability of areas for reintroducing the Bearded Vulture in Andalusia, Southern Spain, potential breeding sites were modelled. The GLM model selected four variables: altitude, topographic irregularity, distance to village and distance to the nearest neighbouring breeding pair. A GIS was used to obtain digital models for each variable. We validated the GLM model testing it with new nest sites in a different geographical context, without including distance to neighbouring pairs. According to the model, in Andalusia the areas with probability > 0.8 coincided with the main mountain ranges. Of these, we selected only protected areas within the historical range of the species. Thus, eight protected areas were considered and surveyed to evaluate their feasibility for reintroduction. We gathered information at two geographical scales: 1-km and 15-km radius around high-probability cliffs. We summarised environmental information on habitat quality and threat level in a matrix that made it possible to compare the viability of reintroduction in each of these eight areas. Poisoning cases and the density of power lines were the most restrictive variables. Sierra Nevada National Park, which was expected to be one of the best areas, showed a high level of threat because of power lines, cable cars and outdoor activities. These threats, however, could be easily mitigated by implementing reintroduction actions. Two protected areas, the Cazorla, Segura y Las Villas Natural Park and the Alhama, Tejeda y Almijara Natural Park, were the most suitable areas for a reintroduction.

Keywords: GIS; GLM; *Gypaetus barbatus*; Habitat assessment; Reintroduction; South Spain.

Introduction

The Bearded Vulture Gypaetus barbatus was once a common species in all the Iberian mountain ranges (Hiraldo *et al.* 1979). Until the middle of the 19th century, it was still broadly distributed in the Penibetic Mountains (Hiraldo *et al.* 1979) but by the end of the 1980s the species had become extinct in southern Spain. Intense poisoning of carnivores and raptors was the main cause, although collecting and hunting also contributed to the decline (Hiraldo *et al.* 1979; Chapman & Buck, 1893). Beginning in 1940, the Bearded Vulture became very rare and by the end of the 1950's, all but one population had been already eradicated. Only in the Cazorla Range did this species survive to recent times. Until the 1960s, six breeding pairs inhabited the Cazorla Range, while another one survived in the adjacent Castril Range. The last breeding occurred in 1983 and the last bird was seen in 1986 (Donázar *et al.* 1991).

After the dramatic results of the 1986 Bearded Vulture census in Cazorla, which resulted in the observation of a single individual, the participants in the third "Coordination Meeting for the Bearded Vulture" recommended a feasibility study prior to the implementation of a reintroduction program. The first of these studies focused on the Cazorla Range (Donázar *et al.* 1991) and the results indicated that Sierra de Cazorla, Segura y Las Villas Natural Park could harbour a population of at least 13-15 breeding pairs. It also recommended extending the study to all other mountain ranges in southern Spain. Following this recommendation, an environmental assessment of the Andalusian ranges began at the end of 2001.

Using a mathematical model based on physical, ecological and anthropogenic variables, Donázar *et al.* (1993) demonstrated that the Bearded Vulture prefers certain types of cliffs. These quantitative studies on habitat selection constitute a very useful tool for conservation planning of endangered species (González *et al.* 1990; Donázar *et al.* 1993; Hirzel *et al.* 2004). Using the same procedure, we can homogenise the selection of the best breeding sites in Andalusia and prioritise areas for reintroduction, following the probability that those areas would be selected by the Bearded Vulture. Moreover, these methods remove most subjectivity when selecting areas (Donázar *et al.* 1991; 1993; Gil-Sánchez *et al.* 1996). Thus, this study analyses most Andalusian mountain habitats as potential breeding sites for the Bearded Vulture. Mathematical models help select potential sites and the spatial capacity of each zone; analyses about the availability of livestock and wild ungulate carcasses determine the trophic loading capacity; whilst analyses of anthropogenic factors that are potentially pernicious, and now affect most landscapes in Andalusia, are used to reject the unsuitable areas. This information will allow us to select the best places for the implementation of a release program in Andalusia as recommended by IUCN guidelines for reintroductions.

Materials and Methods

Selection of Areas

We applied the Generalized Linear Model (GLM) used for habitat selection analyses of the Bearded Vulture in the Pyrenees (Donázar *et al.* 1993) to the Andalusian territory. These authors measured 13 variables relating to 111 nest sites. Among these variables, those that significantly predicted the presence of nest sites were: topographic irregularity, altitude, distance to villages and distance between neighbouring breeding pairs. This model was subsequently tested in a different spatial scale (Bustamante, 1996) and in a different geographical context (Donázar *et al.* 1997). The model was useful when applied on small scales, as in Donázar *et al.* (1993) or Donázar *et al.* (1997).

Before applying the model to the Andalusian territory, we tested it by incorporating 36 new nesting sites that appeared in the Pyrenees between 1990 and 2002. We measured all significant variables except the distance to the nearest neighbouring pair, because we intended to predict nest sites independently of the existence of conspecifics (same procedure as in Bustamante, 1996 and Donázar *et al.* 1997). Of the new nest sites, 69.4% were predicted by the model with a probability > 0.5. Of these, 80% were predicted with a probability ? 0.7, hence, the model was considered valid. The variables used to determine the probability of a cliff being selected were the same as for the GLM: altitude, distance to village and topographic irregularity. To obtain digital models of these variables we used Arcview 3.2 and IDRISI Geographical Information Systems (GIS). We removed a peripheral band 2-km inward from the Andalusian border, due to the lack of reliable information for those areas.

For estimating the probability of suitable nesting sites for the Bearded Vulture in Andalusia, we calculated the linear predictor (LP) (see details in Donázar *et al.* 1993), without considering the presence of neighbouring pairs (Donázar *et al.* 1997): LP = -33.93 + (0.090058 * [Topographic irregularity]) + (0.009867 * [Altitude]) - (4.024*10-6 * [Altitude]2) + (0.9451 * ln [Distance village]) + 15.3135. The resulting probability of the function [P = (eLP)/(1+eLP)], represents the probability of a cliff being selected by the Bearded Vulture. To determine which areas have both a high probability to be selected and the presence of cliffs, we overlapped the coverage of probability (?0.8) and slope (>50º). The final result is a map that includes the cliffs with the highest probability of being selected by Bearded Vultures exclusively (Fig. 1). This coverage was subsequently used in combination with socio-economic, infrastructure and environmental coverages in vectorial format. Arcview was also used to generate 1-km-radius buffers around high-probability-cliffs. These buffers or a group of them were considered as survey areas (SAs, n=59) for subsequent fieldwork studies.

The differential latitude of the study areas (Andalusia vs. Pyrenees) determines some environmental differences. It is to be expected that these differences affect the habitat selection pattern of the species. Indeed, in the

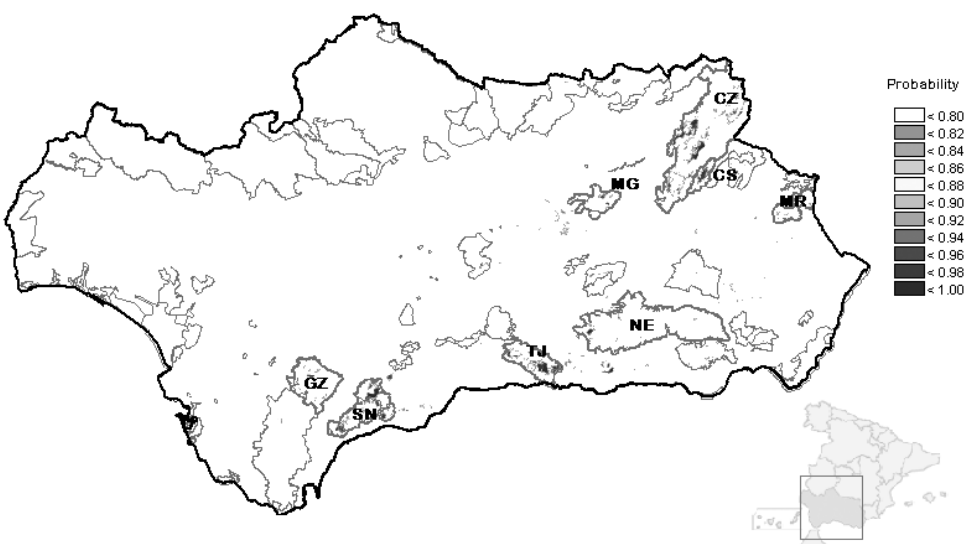

Figure 1. Map of probabilities of a cliff's being selected by the Bearded Vulture *Gypaetus barbatus* in Andalusia. Abbreviations for protected areas are as follows: **CS**: Sierra de Castril; **CZ**: Sierras de Cazorla, Segura & Las Villas; **GZ**: Sierra de Grazalema; **MG**: Sierra de Mágina; **MR**: Sierra de María & Los Vélez; **NE**: Sierra Nevada; **SN**: Sierra de las Nieves; **TJ**: Sierras de Tejeda, Almijara & Alhama.

Cazorla Mountains, the orientation of the cliff and the climatic conditions of the territories are different from those in the Pyrenees (Donázar *et al.* 1991). Moreover, the meridionality of Andalusia would influence the maximum altitude suitable for the Bearded Vulture, which is expected to be higher than in the Pyrenees. There are also differences related to the occupation of territories in lower altitudes. In the Pyrenees, the Bearded Vulture does not select low-altitude territories. This could be the result of the major anthropogenic influence at these elevations. Indeed, in Andalusia, this species inhabited cliffs near sea level (i.e. the Rock of Gibraltar) and in Crete most nesting sites are located below 1,000 m.a.s.l. (Xirouchakis, 2003). In other words, we did not include all suitable altitudes. Nevertheless, for the reintroduction program, the principal goal should be to select large mountainous protected areas with low human influence, and not isolated cliffs, even if they show very good characteristics. In this sense, our model compiles all large mountainous protected areas in Andalusia.

After creating the probability map, we performed a new selection following IUCN guidelines. We selected only those protected areas coinciding with the historical distribution of the Bearded Vulture. In this way, we obtained eight areas belonging to the Andalusian Protected Areas Network (RENPA). Moreover, all of them are also Sites of Community Importance (SCI) and Special Protection Areas (SPA). Other zones not coinciding with these criteria were temporally discarded for further analyses. The eight selected protected areas (PAs), were the object of further analysis (Fig. 1, Table 2, 3). Moreover, most Bearded Vulture sightings (76-95%) occur in an area of 300-700 km^2. In other words, this main foraging area is comparable to a circular area with a radius of 9-15 km. (Brown, 1988) or with a 13.8 km radius (Margalida & Bertrán, 1997). Hence, we selected 15-km-radius zones around the best cliffs selected by the model as second scale of analyses as for the habitat selection analyses in the Pyrenees (Donázar *et al.* 1993). This second scale of analysis results in the inclusion of literally all the surfaces of the PAs selected, and we therefore used the PA scale in order to facilitate the compilation of environmental information and the comparability of PAs.

Table 1. Livestock census (year 2002), removed specimens and annual removal rate, wildlife carcases, mainly mountain goat (Capra pyrenaica), and available biomass. The trophic capacity of the environment is always considerably higher than the spatial capacity of the environment. Abbreviations for protected areas (PA) are as follows, **CS**: Sierra de Castril; **CZ**: Sierras de Cazorla, Segura & Las Villas; **GZ**: Sierra de Grazalema; **MG**: Sierra de Mágina; **MR**: Sierra de María & Los Vélez; NE: Sierra Nevada; **SN**: Sierra de las Nieves; **TJ**: Sierras de Tejeda, Almijara & Alhama.

PA	Livestock (n)	Removed carcasses (n)	Annual removal rate (n)	Livestock (Kg)	Wild ungulates (n)	Wild ungulates (Kg)	Biomass (Kg)	Trophic capacity (pairs)	Spatial capacity (pairs)
NE	120018	692	1038	9534	239	1673	11207	32	5
CZ	141379	699	1049	12453	690	4830	17283	49.4	12
TJ	45027	134	201	4900	200	1400	6300	18	4
GZ	80592	72	108	10528	65	455	10983	31.4	2
SN	15154	45	67.5	1652	10	70	1722	4.9	1
MA	79112	528	792	5530	-	-	5530	15.8	2
MG	29446	0	0	4123	-	-	4123	11.8	3
CS	73835	699	1049	3000	-	-	3000	8.6	1

Table 2. Environmental variables used as indicators of viability for selecting protected areas suitable for the reintroduction of the Bearded Vulture in Andalusia.

VARIABLES CONSIDERED FOR THE VIABILITY MATRIX
Threats
1. Poison. Number of poisoning cases/1,000 ha of the PA. This value is converted by a 3x factor since poisoning is the main cause of mortality in Spain. Information about the cases of poisoning in Andalusia was provided by the Centre for Analyses and Diagnosis of the Andalusia Ministry of the Environment for the period 2001-2004. The information is given as cases of poisoning by each Municipality belonging to the PA during the given period. 2. Power lines. Density of power lines in the PA. This is measured as the length of power lines in the entire PA (m)/extension of the PA (ha), plus the length of lines in the buffer zones (m)/extension of buffer zones (ha). The resulting value (m/ha) includes both the buffer zones and the PA. In this way, we tried to eliminate the uncertainty of the risk of collision outside the buffer zones (potential release zones) during the first movement of release of the juveniles. For example, some buffer zones do not have power lines but instead are very close to them. Moreover, this helps equilibrate the discordance in density of power lines among buffer zones and PA (considered to be the main foraging area). The value is converted by a 2x factor since this is the second highest cause of mortality in Spain. 3. Cable cars. Presence of cable cars. We do not measure the length of cable cars because they only exist in one of the areas. We do not transform this variable due to the slight significance of this factor in Andalusia. 4. Paved roads. Density of paved roads inside the PA, as length of paved road of the PA (m)/extension of the PA (ha). 5. Unpaved roads. Density of unpaved roads inside the buffer zones, as length of unpaved roads of the buffer zones (m)/extension of the buffer zone (ha). 6. Population. Population density, as the number of inhabitants by municipality belonging to the PA/extension of the PA (ha). 7. Outdoors activities. Number of outdoor activities in each AP. For example: climbing, biking, trekking, etc.
Habitat quality
8. Spatial capacity. Number of potential breeding pairs calculated for each PA. 9. Trophic capacity. Number of potential breeding pairs based on food availability. 10. Open habitats. Percentage of the PA with open habitats. 11. Cliffs. Percentage of the PA with cliffs selected by the model. 12. Villages. Absence (1) or presence (0) of villages inside the PA. 13. Griffon vulture. Number of breeding pairs/100 ha of the PA (data up to 2002). 14. Egyptian vulture. Presence of breeding pairs in the PA (1), absence (0) (data up to 2005). We do not include number of pairs/ha because this species inhabits only two PAs and the numbers of pairs is very low. 15. Protected surface. This variable indicates the extension of protected area that could be used by the Bearded Vulture as foraging area. 16. Protection. Degree of protection: Natural Park (1) permits more human activities; National Park (2) has more restricted use, more protection more facility for actions and reduction of threats. 17. Education. Environmental education about the reintroduction program is a determinant factor for the success of the reintroduction program. We evaluated the degree of the viability of the areas following the intensity of education campaigns carried out by the Gypaetus Foundation: absent (0), once (1), continuous (2).

Loading Capacity

We calculated the number of potential breeding pairs likely to inhabit each PA. To do this, we used the information related to the distribution of most suitable cliffs following the model and the mean distance between Pyrenean breeding pairs, estimated to be 11 km (Donázar *et al.* 1993). Donázar *et al.* (1993) stated that their model was not applicable to a location with limited food availability. Hence, we evaluated whether food is a limiting factor or not. We gathered livestock census information from all municipalities belonging to the selected PAs. This information was provided by Agriculture and Livestock Local Offices from each Municipality (OCA). The numbers of livestock correspond to the year 2002. We only used data related to sheep and goats, in accordance with the preferences of the Bearded Vulture (Hiraldo *et al.* 1979; Margalida *et al.* 2001). The Public Company for Agriculture and Fishing Development (DAP) is in charge of removing livestock carcasses from the countryside (following national legislation). They provided information for the period 1 January 2002 to 31 August 2002 (eight months) for the municipalities of the PAs. Company personnel confirmed that many carcasses are not removed because of inaccessibility to the areas or because they do not receive notification of death. Nevertheless, we calculated an annual removal rate from the monthly removal mean. Although the data was of limited value, this information indicated how many animals are removed and how many remained in the countryside. The only PA for which there was no information available was the Castril Range Natural Park. Acting conservatively, we attributed the maximum value of removal to this area, which coincides with the adjacent PA of Cazorla. When available, we included the information on wild ungulates provided by PA administrators (see Table 1). The trophic loading capacity was calculated assuming that the Bearded Vulture can consume up to 7 kg from each medium-sized ungulate carcass (Clouet, 1984; Canut, *et al.* 1987) (see Table 1). Moreover, the annual requirements of a breeding pair are 350 kg (Hiraldo *et al.* 1979) or, in other words, 50 carcasses/year. In general the information on food availability is very conservative because of the following facts: (1) we assumed a 2% mortality rate, although mortality rates are commonly much higher, from 4-8% (Del Junco & Barcell, 1997); (2) we did not consider livestock carcasses other than sheep and goat; (3) livestock censuses do not include juveniles; (4) the availability of wild ungulates is based only on selective hunting and does not include natural mortality. Moreover, for some species (such as wild boar or red deer) there were no data about hunting or censuses for most areas; (5) the trophic diversity of the Bearded Vulture is considerably higher than has been herein assumed (Margalida & Bertrán, 1997; Margalida *et al.* 2001).

Environmental Viability Matrix

We selected the main factor affecting the distribution of territories, productivity, mortality and conservation of the Bearded Vulture (Hiraldo *et al.* 1979; Donázar *et al.* 1991; Donázar *et al.* 1993; Bustamante, 1996; Báguena *et al.* 2004; Hirzel *et al.* 2004) as variables for comparing the eight PAs selected (Table 2). This comparison allows us to determine the main problems and benefits of each area for a reintroduction program. Moreover, the matrix information serves as a guideline for improving the current situation in some of the selected areas (See Tables 2, 3). For comparisons between areas, the values of the different variables were transformed from their original value to 0-1 (Table 3), with the maximum value of the variable equal to 1. Variables were weighted (2x, 3x) in relation to their major implications in the mortality or conservation of the species (see Table 2 for further explanation). The sum of threats is considered the “Threat index” and receives a negative value. The “Habitat quality” index has a positive value, and is the sum of all variables positively affecting the survival of the species. The difference between both indexes results in the “Viability index” (Table 3).

Table 3. Viability matrix representing all selected protected areas and their compared values of threat and habitat quality. The viability index is the sum of both habitat quality index and threat index. In parenthesis is the original value of the variables before transformation. Abbreviations for protected areas are as follows, **CS**: Sierra de Castril; **CZ**: Sierras de Cazorla, Segura & Las Villas; **GZ**: Sierra de Grazalema; **MG**: Sierra de Mágina; **MR**: Sierra de María & Los Vélez; **NE**: Sierra Nevada; **SN**: Sierra de las Nieves; **TJ**: Sierras de Tejeda, Almijara & Alhama.

	Protected areas							
Threats	**CZ**	**CS**	**MG**	**NE**	**TJ**	**GZ**	**SN**	**MR**
1. Poison	0.10 (0.04)	0.80 (0.24)	1.29 (0.40)	0.33 (0.10)	0.20 (0.05)	1.53 (0.48)	3.00 (0.94)	0.30 (0.09)
2. Power lines	1.76 (3.26)	1.10 (2.05)	0.13 (0.47)	2.00 (3.76)	0.30 (0.50)	1.40 (2.64)	0.38 (0.70)	0.38 (0.71)
3. Car cables	0(0)	0(0)	0(0)	1(1)	0(0)	0(0)	0(0)	0(0)
4. Paved roads	0.78 (2.29)	0 (0)	0.60 (1.72)	0.33 (0.96)	0.10 (0.19)	1 (2.93)	0 (0)	0.22 (0.64)
5. Unpaved roads	0.48 (7.20)	0.2 (2.80)	0.47 (7.10)	0.64 (9.60)	0.40 (5.40)	0.35 (5.35)	0.60 (9.06)	1 (15.09)
6. Population	0.20 (0.41)	0.10 (0.26)	0.81 (1.80)	0.21 (0.47)	0.40 (0.98)	0.64 (1.43)	1 (2.23)	0.12 (0.26)
7. Outdoor activities	0.80 (4)	0.80 (4)	0.40 (2)	1 (5)	0.80 (4)	0.80 (4)	0.80 (4)	0.80 (4)
Habitat quality								
8. Spatial capacity	1 (12)	0.10 (1)	0.25 (3)	0.42 (5)	0.30 (4)	0.17 (2)	0.08 (1)	0.17 (2)
9. Trophic capacity	1 (49.4)	0.20 (8.60)	0.24 (11.8)	0.65 (32)	0.40 (18)	0.64 (31.4)	0.10 (4.90)	0.32 (15.80)
10. Open habitats	0.34 (0.17)	0.88 (0.44)	0.48 (0.24)	1 (0.50)	1 (0.50)	0.60 (0.30)	0.80 (0.40)	0.60 (0.30)
11. Cliffs	0.50 (0.31)	0.76 (0.47)	0.4 (0.24)	0.10 (0.06)	1 (0.62)	0.16 (0.10)	0.52 (0.32)	0.11 (0.07)
12. Villages	0(0)	1(1)	0(0)	0(0)	1(1)	0(0)	0(0)	1(1)
13. Griffon vulture	0.20 (0.14)	0.35 (0.24)	0 (0)	0 (0)	0 (0)	1 (0.69)	0 (0)	0.12 (0.08)
14. Egyptian vulture	1(4)	0(0)	0(0)	0(0)	0(0)	1(7)	0(0)	0(0)
15. Protected surface	1 (214300)	0.10 (12265)	0.09 (19961)	0.80 (171986)	0.20 (40663)	0.24 (51695)	0.09 (20163)	0.10 (22670)
16. Protection	1(1)	1(1)	1(1)	2(2)	1(1)	1(1)	1(1)	1(1)
17. Education	2(2)	0(0)	1(1)	1(1)	0(0)	0(0)	1(1)	0(0)
Habitat quality Index	**8.04**	**4.39**	**3.46**	**5.97**	**5.90**	**4.81**	**3.59**	**3.42**
Threat Index	**-4.12**	**-3.0**	**-3.7**	**-5.51**	**-2.2**	**-5.72**	**-5.78**	**-2.82**
Viability Index	**3.92**	**1.39**	**-0.2**	**0.46**	**3.7**	**-0.91**	**-2.19**	**0.6**

Results

From a geographical point of view, the distribution of high-probability cliffs is related to the main mountain ranges south of the Guadalquivir River or in its headwaters (Fig. 1). These areas also coincide with most historical geographical records for the species (Hiraldo *et al.* 1979). According to the viability matrix, the evaluation of the areas would be as follows (see Table 3).

In the southwest, Sierra de las Nieves (SN) and Grazalema Natural Parks (GZ) show very high threat index, mainly due to poisoning. But, while Sierra de las Nieves has low habitat quality (and small size), Grazalema possess good habitat quality. For example, trophic and spatial capacity, open habitats and the existence of breeding pairs of Griffon and Egyptian vultures certify the environmental viability of this area for the Bearded Vulture.

In eastern Andalusia, Sierra Mágina and Sierra de María are two small areas potentially suitable. The habitat quality of Mágina is not very high, in relation to the rest of the areas, mainly due to the scarce spatial capacity, small size, scarce amount of open habitats and the presence of villages inside the PA. But the main negative factor is poisoning. On the other hand, Sierra de María & Los Velez, shows good quality index, but the small size of the protected area led us to discard it for an initial reintroduction.

Sierra Nevada Natural and National Parks (broadly overlapped) shows a moderately low viability index. This is due to its high threat index, although poison seems to be scarce. This low viability index strongly contrasts with the very high habitat quality index, the second of all selected PAs. Nevertheless, the high density of power lines, cable cars, the intensity of outdoor sports and the high density of roads near the buffer zones result in a strong negative threat index. Fortunately, the threats affecting this PA are easily manageable.

In Sierra de Cazorla, Segura & Las Villas Natural Park the environmental conditions are the best of the PAs studied. The high trophic and spatial capacity, the presence of Griffon and Egyptian vultures, the amount of protected lands and the low population density makes this area suitable for the Bearded Vulture. Moreover, poison, the major general threat, has the lowest value in this area. Nevertheless, the threat index is still high due to power lines, paved roads, other roads and outdoor sports. The paved roads inside the Park are far from buffer zones and thus are not a potential factor for breeding pairs.

Sierra de Castril Natural Park is a small PA adjacent to the Cazorla PA and has the same ecological conditions. The viability index is high, mainly due to a scarce level of threats. Its primary negative factor is power lines. The spatial and trophic capacity is not a determining factor because it belongs to the same geological entity as Cazorla, Segura y Las Villas Range, forming a continuous zone that is only separated by administrative provincial borders.

Sierras de Alhama, Almijara y Tejeda Natural Park is the second most important area. The low number of cases of poisoning and low density of power lines, paved and unpaved roads, give this PA the lowest threat index. Although the population density of the municipalities forming the Park is high, the area shows a very low influence from the surrounding villages, at least in terms of affecting the Bearded Vulture. Moreover, there are no populated areas inside the Park. The very high density of cliffs and open areas makes this area one of the better places for the Bearded Vulture in Andalusia.

Discussion

There is a broad spatial coincidence between the areas selected by the model, the historical distribution of the Bearded Vulture (Sánchez-García, 1885; Valverde, 1956; Hiraldo *et al.* 1979; González & González, 1984; Pleguezuelos, 1991; Donázar *et al.* 1991) and some Andalusian PAs. These parks are also included in the European strategies for nature conservation (Natura 2000 Network). Only eight of these parks meet the necessary conditions (area selected by the model, protected area and historical presence). Fieldwork in these areas revealed a high availability of cliffs. Food was also not a limiting factor, widely surpassing the spatial capacity (Table 1). Nevertheless, some of the eight selected PAs do not meet habitat or threat conditions to ensure a successful reintroduction program. Therefore, we selected those with the best habitat conditions and least degree of threat. Moreover, we considered those areas with enough extension to assure the protection and the establishment of a population. The most suitable areas are Cazorla, Segura y Las Villas, Tejeda-Almijara, Castril and Sierra Nevada. In the first three areas, reintroduction could be initiated in the short term, after completing the modifications and indication of power lines, but Sierra Nevada, despite its very high quality index, requires more detailed analysis and management.

Hiraldo *et al.* (1979) summarised in detail the dramatic decrease in all Spanish populations of the Bearded Vulture. Poisoning was without any doubt the main cause of decline for the species in the Iberian Peninsula. Already at the end of the 19th century and the beginning of the 20th, authors such as Chapman & Buck (1983, 1910) had commented on the negative impact that poisoning used against carnivores, like wolves and foxes, was having on Bearded Vulture populations. During the first three quarters of the 20th century, massive poisoning was a common practice across Spain (i.e. Hiraldo *et al.* 1979; Pleguezuelos, 1991; Donázar *et al.* 1991; Hernández *et al.* 2001). Moreover, this practice was paid quite well after the foundation of "Vermin Extinction Committees" that were in charge of eradicating all potential predators (Donázar *et al.* 1991). Hunters, forestry services, shepherds, village inhabitants, etc., used poison, mainly strychnine, to reduce the carnivore community. Unfortunately, poisoning is still widely practiced in Europe, from Spain to the UK, Austria, France, Italy and Greece (i.e., BVCF, 2000; Barnett *et al.* 2004; Gerstl & Böck, 2005).

Poisoning is illegal and strongly persecuted in Andalusia, although it is still common (Hernández *et al.* 2001; Couto *et al.* 2005). But, although poisoning is the major threat, the current situation is not comparable with past massive, legal and popular poisoning campaigns that led to the extermination of a considerable number of raptor populations. Poisoning impact on wildlife has been reduced considerably and raptor populations are increasing or, at least, are stable in areas of historical decline (i.e., Moleón *et al.* 2004). Even species such as the Griffon Vulture, which were eradicated from the Granada province (Andalusia) through intensive poisoning (Pleguezuelos, 1991), are now undergoing a notable recovery process (Donázar & Fernández, 1990; Moleón *et al.* 2004). Some species whose populations were dramatically reduced (i.e. *Aquila adalberti* and *Neophron percnopterus*) are confronting serious survival problems, probably also due to several facts related to dyamics of small populations. In fact the Black Vulture is undergoing a constant population increase in Andalusia (CMA-Junta de Andalucía, 2001) despite its vulnerability to poisoning. Nevertheless, a recent study (Couto *et al.* 2005) reveals that, in Andalusia, there is no data to determine whether the increase in poisoning detected in recent times is the result of an increase in poisoning activities or it is caused by more intensified searching, increased public awareness, more specific information to Environmental Agents and the advance in detection techniques (such as specifically trained dogs). But, undoubtedly, poisoning is still a serious problem (BVCF, 2000; Báguena *et al.* 2004). For example, in the Pyrenees, this activity caused 62% of the mortality from 1994 to 2002. Hence, it is currently the most important non-natural mortality factor for the Iberian population (Báguena *et al.* 2004) although the current effect seems to be lower than in historical times (Heredia & Heredia, 1991) and the species now has a more hopeful future (Báguena *et al.* 2004).

Moreover, poison is still present in other areas where reintroduction programs have been successfully implemented (i.e. the reintroduction of the Red Kite *Milvus milvus* in Britain or the Bearded Vulture in Austria and Switzerland). It must also be borne in mind that the Andalusian Government was carrying out during the last years probably the Europe's strongest action plan against poisoning, with an annual budget of 300.000 €. Poison baits are searched by specifically trained dogs with a 65% success rate (Couto, *et al.* 2005), and by specifically equipped and trained environmental agents. Therefore, the number of cases may seem disproportionately high in comparison with areas where no specific searches are being carried out at all. Thus, the lack of the same or similar methods and equivalent sampling efforts does not permit direct and objective comparisons between most areas.

The Andalusian Threatened Species Recovery Centres (CREA) provided all the information about the number of raptors received in those centres because of illegal shooting between 1999 and 2001. Since only four cases were detected for all the protected areas under consideration and the information was scarce and scattered, we decided not to include this data in the viability matrix. Certainly shooting has more impact on raptors than is reflected in the data provided by recovery centres and this fact should be studied carefully. Indeed, deaths of Bearded Vultures by shooting represent 10.3% of the mortality in the Pyrenees (Báguena *et al.* 2004). Illegal hunting is, therefore, a problem that needs to be monitored more closely in the future.

Although historically power lines were probably not a main cause of decline for the Bearded Vulture, they currently pose a threat (Fernández & Azcona, 2002; Báguena *et al.* 2004). According to Báguena *et al.* (2004), in the Pyrenees electrocution and collision represented 6.8% and 3.4% of the mortality causes respectively (from 1994 to 2002). Since juveniles tend to use the release zones more frequently during the time right after their release, sites selected for release need to be free of power lines, or those lines must represent no threat to the individuals. Moreover, since the PA would be the area most regularly used for foraging by the Bearded Vulture, at least during the first period, a more complete analysis of the PA's power lines should be carried out before release.

We recommend extending the viability studies to other non-protected mountain ranges potentially suitable for the Bearded Vulture, in order to improve our knowledge of the potential dispersion area. The geographical connection between the Cazorla, Segura y Las Villas Range and the Castril Range and the adjacent Ranges of Murcia and Albacete makes it also necessary to extend the viability studies to those areas.

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POPULATION VIABILITY ANALYSIS OF (EURASIAN) GRIFFON VULTURE *Gyps fulvus* IN CROATIA

Gordana Pavokovic[1] & Goran Susic[2]

[1] Eco-center "Caput Insulae-Beli", Beli 4, Island of Cres, Croatia
E-mail: gogapa@inet.hr
[2] Ornithological Station Cres, CASA, Beli 4, Island of Cres, Croatia
E-mail: orlov-let@ri.t-com.hr

Abstract

Vortex, a simulation modelling programme, was used to explore the effects of deterministic forces as well as demographic and environmental stochastic events on a Griffon Vulture population in Croatia. Model construction was based on data from field surveys performed monthly (from 1995 to 2004) on the Island of Cres and other Kvarner islands and on data on resightings and recoveries of marked/ringed birds (from 1990 to 2003).

Vortex is a heuristic tool, not a predictive one. It is a tool for exploring "what if" questions, comparing management strategies and generating hypotheses. It projects stochastically the interactions of the many parameters that enter into the model. This population model includes variability in survival rates for four age classes, reproductive rates, demographic stochasticity, carrying capacity and effects of changes in mortality and reproduction rates. The population model was subjected to a basic scenario, using the most realistic data and five different scenarios that examined catastrophes. Two of these scenarios were examined in two versions, with 10% and 30% frequencies of catastrophic events. The probability of population decline within the next 25 and 50 years was assessed. The model appeared to make reasonable predictions for the Griffon Vulture population. Generally, the simulation results showed that the risk of extinction greatly depends on the survival rate, reductions in the population growth rate and frequencies of catastrophes.

Keywords: Population Viability Analysis; Griffon Vulture; Vortex; Croatia.

Introduction

Amongst the Adriatic islands, the Kvarner islands are home to the greatest biodiversity. Plant and animal species richness has been maintained primarily due to the specific requirements of the human population living on the island. The traditional practices of sheep raising, retained for thousands of years, has enabled proper maintenance of plant and animal species and thereby has promoted the survival of Griffon Vultures *Gyps fulvus*, which is linked to the practice of leaving sheep carcasses in the field.

The Griffon Vulture is the only one of the four European vulture species that inhabits Croatia. At the end of 19th century these vultures inhabited large areas of the country, particularly along the Dalmatian coast and throughout the Adriatic islands. At the beginning of the 20th century, the Croatian Griffon population strongly declined and today they only nest on the Kvarner islands of Cres, Krk, Plavnik, Prvic, and occasionally, the Island of Pag (Susic & Grbac, 2002).

Currently, less than 90 pairs inhabit Croatia. Of this, approximately 65 pairs or 80% exist solely on the Island of Cres. However, even this colony has entered a declining phase. Between 1995 and 2001, a decline from 25 to 15 pairs has occurred on the Island of Prvic, whilst on the Island of Krk, a fall from 12 to 6 pairs over 5 years has been recorded. The Griffon Vulture is peculiar to the coastal area of Croatia as it breeds exclusively on the cliffs above the sea. At the end of September all juvenile birds leave the Croatian colonies and migrate. They use three main directions: north-west to the Austrian and Italian Alps, then France and Spain; south-west to the southern Italy and south-east to Bulgaria, Greece and Turkey, to Israel and as far south as Chad (Susic & Grbac, 2002). Unfortunately, illegal poisoning and the disappearance of the extensive traditional sheep farming in these breeding areas, as well as human disturbance (as tourism or recreation) near nest sites in the coastal zone of Croatia, threaten their survival. This remnant population is also particularly vulnerable due to low immigration rates.

Vortex computer programme is a simulation of the effects of deterministic forces, as well as demographic, environmental and genetic stochastic events on wild animal populations (Miller & Lacy, 2003). It models population dynamics as a separate series of events which appear according to the probabilities generated by certain random variables which follow distributions specified by the user of the programme. Vortex simulates population through a series of events that describe the annual cycle of a typical diploid organism with sexual reproduction: selection of partner, reproduction, mortality, growth, migration, emigration and immigration. Vortex creates the image of every individual within its memory and follows its fate throughout its life. It simulates processes of birth, death and gene transition through generations, producing pseudorandom numbers which will determine whether each individual will live or die, which adults will mate, who will produce offspring, what will be their sex. Demographic events (birth, sex, reproduction, distribution and death) are modelled for each animal in every single year of the simulation when a particular event had happened. Vortex requests much specific data about the population. We used this simulated model to the Kvarner population of Griffon Vultures.

Materials and Methods

Simulation of the Kvarner Griffon Vulture Population

The initial population size of N individuals will be different after $t = T$ years if initial population size is bigger and if simulation period is longer. To test it, initial population size is simulated for $t = T$ years using 500 repetitions. For every set of repetitions we make risk curves and they together create a contour map, showing the probability of the final population size. Each figure shows the probability according to the initial population size for a given time period. Number of years varies

Table 1. Assumed age distribution for the population size of 146 individuals.

Age/year	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	
N male	18	4	4	3	4	4	4	3	3	3	3	3	3	3	3	2	2	2	1	1	73
N female	18	4	3	4	3	4	4	3	3	3	3	3	3	3	3	2	2	2	1	1	73

from 25 to 50 in 5 year steps. We used Vortex version 9.51 (Lacy *et al*. 2004).

Risk is defined as the probability of an undesirable event (such as population extinction or population decline). Estimation of the extinction risk is an attempt to foresee the probability of population (or species) extinction or decline through certain time period under different natural events and scenarios of population management (Akçakaya, 2000).

Parameters Used in Modelling with VORTEX

Parameter estimations come from estimation of reproduction and survival rates and from scientific papers and other publications on Griffon Vultures. Demographic parameters used are average values for the population in the period 1990-2004, as well as from published data on Griffon and other Gyps vultures. Extinction is defined as a lack of individuals of one or both sexes. ***As Initial population size N*** we used 146, which is the mean of an estimation of the total population on the Island of Cres in the period 2000-2004. Age-class distribution for these 146 individuals is shown in Table 1. It is a synthesized age distribution equivalent to a stable age distribution of the life table matrix, presuming an annual rate of population change (λ = 1). Initial ***Carrying Capacity, K*** = 1000. *In VORTEX K* stands for carrying capacity and it means the size of population above which all individuals could die (Nilsson, 2004). Real carrying capacity is probably even bigger, taking into account former population size. In *VORTEX*, if the population size exceeds *K* at the end of particular time cycle, additional mortality is imposed across all age classes in order to reduce the population back to this upper limit (Miller & Lacy, 2003). ***Extinction*** is defined as the absence of one or both sexes in the population. ***Inbreeding depression*** is modelled with suggested 3.14 lethal equivalents (inbreeding coefficient) per individual.

VORTEX assumes that an animal can constantly breed during adulthood. **Maximum age of reproduction** is estimated at 20 years, because average longevity is 22.41 to 32.86 years for wild Griffons (Piper, 1994). Hatzofe (pers. comm.) states 33-35 years for Griffons in captivity. Mundy (2000) stated that in 1994 a 19 year-old dead ringed Cape Vulture *Gyps coprotheres* was found. The Spanish Imperial Eagle *Aquila adalberti*, with similar demographic characteristics as Griffon Vulture, can reach the age of 21-22 years in the wild (Ferrer *et al*. 2004).

VORTEX DEFINES breeding as the time when the first offspring are born, not the age of onset of sexual maturity. In this paper it is assumed that age of first reproduction is the same for males and females, and is taken as 4 years, based on research by Sarrazin (1996). **Sex ratio at birth** is 50:50, all adult males potentially can reproduce, but in fact the number of reproducing males depends on the number of reproducing females (and vice versa). The percentage of females which reproduce, in relation to the total number of females, is 60 (the same value as with breeding success). The **reproductive system** is long-term monogamy because Griffon Vultures stay with their mate for many years. **Standard deviation of environmental variation** in percentage of adult females which breed is 6.6. **Maximum number of progeny per year** is one; every reproductive female lays one egg per year. The reproductive cycle starts at the time of egg laying, which in this area starts at

Table 2. Mortality rates (5%) for Griffon Vultures used in *VORTEX*.

Age classes	Mortality rates (%) Females	SD due to environmental variation	Mortality rates (%) Males	SD due to environmental variation
0-1	24	26	24	26
1-2	11	27	11	27
2-3	3	29	3	29
3-4	1	27	1	27
4 and older	1	27	1	27

the beginning of the calendar year. Adult survival is counted from the age of 4 when females reach sexual maturity. There is no reason to separate age of sexual maturity between males and females. The programme MARK (White & Burnham, 1999) was used to obtain survival rates of marked birds (Pavokovic, 2006). The values of mortality rates are shown in Table 2. No difference in mortality rates between males and females was considered, i.e. the average value for both sexes was entered into the model.

The populations were projected in two time periods, 25 and 50 years. Each projection was run 500 times, as recommended by Harris *et al*. (1987), providing probabilities of extinction and population sizes with standard errors 2% of the predicted values. Time periods were chosen to present: a period in which a strategy of preserving the species could be implemented and tested (25 years) and a time period (50 years), which could represent the "long term" survival of the population. Simulation results are summed in terms of probability over a given time horizon.

Catastrophes are single, extraordinary events outside the borders of normal environmental variations, which alone, or in combination with other factors, affect reproduction and survival. Examples of nature catastrophes are drought, disease, lack of prey, fires, poisoning, or combinations of these events. Catastrophes are modelled by the probability of occurrence and with a factor of severity, which may vary from 0.0 (maximum or absolute effect) to 1.0 (no effect).

Scenarios Used in Simulations

The average population growth rate after 500 simulations and the probability of extinction are compared through six different scenarios. The basic scenario uses the most realistic data. The additional five scenarios are based on realistic options of population management or potential threats. Each scenario is defined as a change in environment or population management and changes are based on the current situation. Two scenarios (*Veterinarian drug* and *Pollutant*) are presented in two versions, with 10% and 30% frequencies of catastrophe occurrence. A summary of scenarios is presented in Table 3.

1. *The Basic scenario*. Realistic data were used for the population status. In this scenario, catastrophes were not used, since simulations with catastrophes assume that such a catastrophe did not happen in the period when the demographic data was collected. If the data did include a catastrophe, then the estimates of demographic

Table 3. Scenarios used in population viability analysis for the Griffon Vulture.

	Name of the scenario	Mortality	Fecundity	Catastrophe	Frequency	Effect on reproduction	Effect on survival
1.	*Basic*	Used real data		No	-	-	-
2.	*The most optimistic*	Decreased 20% in relation to Basic scenario	Increased 20% in relation to Basic scenario	No	-	-	-
3.	*Scarce food*	Increased 10% in relation to Basic scenario	Decreased 10% in relation to Basic scenario	Yes	30%	0.80	0.70
4.	*Massive poisoning*	Same as in Basic scenario	Same as in Basic scenario	Yes	20%	0.95	0.80
5.	*Veterinary drug*	Same as in Basic scenario	Same as in Basic scenario	Yes	10%	1	0.50
6.	*Pollutant*	Same as in Basic scenario	Same as in Basic scenario	Yes	10%	0.50	1
5.a	*Veterinary drug f30*	Same as in Basic scenario	Same as in Basic scenario	Yes	30%	1	0.50
6.a	*Pollutant f30*	Same as in Basic scenario	Same as in Basic scenario	Yes	30%	0.50	1

rates should exclude those catastrophe years. Otherwise, the combined effect of modelled catastrophes and environmental stochasticity would overestimate the actual variability in vital rates and underestimate their means (Akçakaya, 2000). The model is unrealistic because it assumes that in the future there will be no changes in parameters referring to demography, limitation in number of available mates, catastrophes, or any other changes in environment.

2. *The most optimistic scenario* - low mortality, high breeding success. It is assumed that increased additional feeding at vulture restaurants will have a positive impact on demographic parameters, especially on decrease of mortality rate by 20% in relation to the Basic scenario for all age classes and a 20% increase in breeding success

(from 60% to 72%). In this scenario, catastrophes are not included; other parameters are the same one as in the Basic scenario.

3. *Scarce food* – The most pessimistic scenario in which we assume that food is scarce due to low numbers of sheep, as a consequence of an increase in alien invasive game species on Kvarner islands and a lack of supplementary food at the feeding stations. The scenario is hypothetic and designed to show limitations in reproduction and survival rates. Mortality is shown as 10% higher than the one in the *Basic scenario* for all age classes (m1 from 24% to 26.4%; m2 from 11% to 12.1%; m3 from 3% to 3.3%; m4 from 1% to 1.05%), and breeding success is 10% lower (from 60% to 54%) than at the *Basic scenario* in all time projections. The *"Scarce food catastrophe"* for Griffons has a frequency of 30% (very pessimistic) in the projected time series (25 and 50 years). The severity factor with respect to reproduction is 0.8, meaning that in the year of catastrophe 20% less females will breed. If normal productivity in the described scenario is 54%, then only (54%) x (0,8) = 43.2% adult females breed in a year with catastrophe. The severity factor on survival is entered as 0.70 (VORTEX here counts survival for all age classes within the population, i.e. survival of juveniles at the *Basic scenario* is 76%, in scenario *Scarce food* survival is decreased for 10%, and in the year of catastrophe it is additionally decreased for 30% (68.4%) x (0.7) = 47.88%. All other parameters of the model are the same as those in the *Basic scenario*.
4. *Massive poisoning* – The catastrophe modelled in this scenario is when a part of the population dies due to ingestion of poison (e.g. such an incident happened on the island of Rab in December 2004, when 17 Griffon Vultures died after consuming a carcass with Carbofuran). The victims represented 10% of the population. This scenario is considered as realistic because many individuals may die from one poisoned bait. Mundy (2000) describes a case of poisoning when Carbofuran was placed on a donkey carcass at a Namibian National Park in 1995 and 86 Lapped-faced Vultures *Torgos tracheliotos* and 8 White-backed Vultures *Pseudogyps africanus* died. In the model the annual probability of *"massive poisoning"* is 20% in the projected time series. The severity factor with respect to survival is modelled on average at 0.80, which means that in the period of the catastrophe the survival rates will decrease by 20% in each age category of the population, which is quite realistic, and the severity factor with respect to reproduction is 0.95 (which means that this demographic parameter in the time of catastrophe will decrease by 5%, i.e. in a normal period 60% of adult females breed, while in the catastrophe period this proportion would be 57%). Fecundity and mortality rates are the same one as in the *Basic scenario*.
5. *Veterinary drug* – It is well known that a 22-50% annual decline in *Gyps bengalensis, Gyps indicus* and *Gyps tenuirostris* populations had taken place in India and Pakistan during 2000-03 (Green *et al.* 2004). Studies have shown that the non-steroidal anti-inflammatory drug Diclofenac has been causing renal failure and was lethal for vultures when they fed on carcasses of livestock treated with normal doses of this drug before death (Oaks *et al*. 2004). So far, in Croatia no tests are done on livestock for veterinary drugs and pesticides before they enter the market. So, it is reasonable to assume that some drugs for livestock could enter into the food chain of Griffon Vulture and might have an impact on Kvarner's population. In this scenario it is assumed that the effect of the veterinary drug ingested by Griffons does not affect their reproductive performance (severity factor 1), but has a great impact on survival, which is decreased for 50% compared to its value in the *Basic*

scenario, i.e. severity factor with respect to survival is 0,50. Frequency of occurrence of this catastrophe is 10%.

5a. *Veterinary drug f30* – The scenario settings are the same as in *Veterinary drug*, but the frequency of catastrophe becomes 30% in the projected time series.

6. *Pollutant* – It is assumed that pollutant affects the reproduction of Griffon Vultures, or, as in the case of DDT, it reduces the availability of calcium carbonate during egg formation in the body of female, so egg-shells are very thin, fragile, or cause dehydration of embryo due to excess loss of water through the shell (Newton, 1998). The scenario is similar to the *Veterinary drug*, the cause of catastrophe could also be a pesticide or drug the impact of which on wildlife has not been tested and Griffons could acquire it into their bodies. In this scenario the effect on reproduction is 0.50 (a 50% demographic rate decrease during the years of catastrophe), the frequency of catastrophe is 10%, and severity factor for survival is 1 (survival rate is the same one as in the *Basic scenario*).

6a. *Pollutant f30* – The scenario settings are the same as in *Pollutant*, but frequency of catastrophe occurrence is 30%.

Sensitivity Analyses of Demographic Parameters

Sensitivity analyses are important components of modelling. They provide useful information to the user of the model as they emphasize which parameters had the highest influence on the model results. Usually, sensitivity analyses are measured by a small variation of parameters around the assessed values. Parameters within the model were varied one at a time to assess which of them had the biggest effect on model prediction. The resulting change in the given variable renders the sensitivity index of the model for certain parameters (Burgman *et al.* 1993). Sensitivity analyses are useful if we want to know which parameters should be estimated more thoughtfully. For instance, if the risk of population decline is very different with low and high survival rate of adults, then results appear to be sensitive on that certain parameter and it is apparent that future field research should focus on adult survival rates, in order to estimate it more accurately (Akçakaya *et al.* 1999). If we want to know which parameter affects more the model predictions, each parameter within the model must be varied in time t: Mortality rates, female reproduction success, maximum reproduction age, initial population size, are increased or reduced by 10% or certain number of years. Since model results could be very sensitive on carrying capacity K, we have also made a second analysis with half the initial value of K. We compared percentage of change in average population growth rates before the sensitivity model and population growth rate in the *Basic model*.

Results

PVA results have to be interpreted as a logical result of our opinion on functioning of the system (Starfield, 1997). Model predictions are not facts. They are hypothetic scenarios of what could happen in the future, based upon our present knowledge on species. Hypotheses have to be tested in the field and not presented as facts. But, even the best models reflect past experiences and suggest that similar things will happen in future. We cannot say with certainty whether the future will resemble the past: new illnesses could be imported, habitats may change, or species lost. We cannot quantify the future, which remains a combination of the known and the unknown (Rosenhead, 1989). The structure of the model and the way questions are put determine the presentation of results. In most cases, models include random variations (stochastic), meaning that results are presented in probability terms, i.e. terms of risk, possibility and probability. Results of population viability analysis are probabilities of population projections in defined periods of time.

In the *Basic scenario*, the deterministic growth rate is identical for both the projected time series of 25 and 50 years; *r*, the exponential rate of population increase is 0.106; λ, the annual rate of population change, is 1.112; R0, the per generation rate of change or "net replacement rate", is 3.060; the generation time for females and males is 10.53 years. This value is a function of the age at first reproduction, maximum age of reproduction and interval between births of offspring. That means that 9-10 generations of Griffon Vultures are taking place within 100 years.

Summary statistics of each population for each scenario are presented in Table 4.

Table 5 shows the results of sensitivity analysis of the population viability analysis for the Griffon Vulture population. The value of *Basic model* for *r*-exponential rate of population increase is 0.103, which equals a realistic growth rate. The model is most sensitive on parameters such as the maximum age of reproduction and age of first reproduction.

Discussion

The reasons to preserve Griffon Vulture in Croatia are both ethical and aesthetic and by preserving this species the whole ecosystem of the area will be preserved. The Griffon Vulture shares a few characteristics with many other extremely vulnerable species: postponed maturity, cooperative search for food, monogamy [95% of identified pairs stays with partner next year (Sarrazin *et al.* 1996)] and low fecundity (Lacy, 2000). For Griffon Vultures a stable environment is of utmost importance. We use the term stable in a subjective sense, meaning that factors which could cause a catastrophic population decline are removed. Griffons do not have the capacity to increase their population rapidly after a population decline. Another requirement for

Table 4. Summary of the six scenarios with 500 iterations projected on 25 and 50 year time series and the scenarios *Veterinary drug f30* and *Pollutant f30* with higher frequencies of catastrophe occurrence in the projected time series: det-r –deterministic rate of population growth; *stoc-r* – mean stochastic growth rate experienced in simulations; *SD(r)* – standard deviation of stochastic population growth rate; *PE* – probability of extinction; *N-extant* – mean of extant population; *N-all* – mean of all populations; *Mean TE* – mean of time of extinction.

Scenario	det-r	stoc-r	SD(r)	PE	N-extant	SD(Next)	N-all	SD (Nall)	Mean TE
Basic 25	0.106	0.103	0.093	0.196	971.33	75.33	780.95	391.84	13.1
Basic 50	**0.106**	**0.105**	**0.091**	**0.410**	**987.33**	**43.12**	**582.46**	**487.16**	**22.8**
The most optimistic 25	0.138	0.124	0.104	0.194	982.15	59.17	791.61	392.37	12.2
The most optimistic 50	**0.138**	**0.127**	**0.097**	**0.328**	**989.43**	**41.73**	**664.89**	**466.24**	**22.3**
Scarce food 25	-0.011	-0.033	0.204	0.266	114.17	126.11	83.82	119.21	13.4
Scarce food 50	**-0.011**	**-0.034**	**0.203**	**0.566**	**101.41**	**138.13**	**45.12**	**104.81**	**25.4**
Massive poisoning 25	0.061	0.054	0.130	0.264	613.95	279.19	451.86	361.54	12.4
Massive poisoning 50	**0.061**	**0.055**	**0.130**	**0.418**	**886.45**	**168.42**	**515.91**	**456.11**	**22.8**
Veterinary drug 25	0.055	0.030	0.228	0.244	433.45	340.05	327.69	349.41	12.1
Veterinary drug 50	**0.055**	**0.028**	**0.231**	**0.416**	**533.54**	**348.85**	**311.59**	**374.52**	**22.6**
Pollutant 25	0.101	0.098	0.093	0.206	964.30	92.01	765.66	398.90	11.7
Pollutant 50	**0.101**	**0.098**	**0.094**	**0.420**	**980.44**	**54.26**	**568.66**	**486.15**	**22.1**
Veterinary drug 25 f30	-0.056	-0.108	0.342	0.466	49.07	83.97	26.35	65.97	15.6
Veterinary drug 50 f30	**-0.056**	**-0.109**	**0.341**	**0.906**	**37.66**	**121.37**	**3.62**	**38.45**	**24.4**
Pollutant 25 f30	0.089	0.088	0.090	0.220	943.64	121.13	736.04	405.65	12.6
Pollutant 50 f30	**0.089**	**0.088**	**0.091**	**0.352**	**982.57**	**47.38**	**636.70**	**471.28**	**22.8**

the preservation of a population with a low reproduction rate is the absence of mortality caused by predation. Griffon Vultures on Kvarner archipelago do not have natural predators others than man. If persecution of the species is continued, losses would surpass births and cause population decrease (Mertz, 1971). Big scavengers are extremely vulnerable to high mortality, especially of adult birds, so populations drastically decrease when suffering high losses, as in the case of poisoning (Donazar & Fernandez 1990). This has also been shown by the *Massive poisoning* scenario. When persecution stops, population can recover, but *Gyps* populations theoretically cannot increase by more than 3% per year (Piper, 1994). In Spain Donazar & Fernandez (1990) have shown that Griffon Vultures do have the capacity to increase population size if enough food is provided and unnatural causes of death are removed.

However, most models are made of deficient data. Coulson *et al*. (2001) argued that data on certain species have to be of extremely high quality, if we want to make accurate estimations of probability of extinction, and if vital rates are to be estimated correctly in the future. The Kvarner population of Griffon Vultures has been monitored very well over the last 20 years, so data could be considered as of high quality. Estimations of catastrophic events are very uncertain, but they may help to realize actual threats for the population.

Model results such as for extinction risks, or declines of population numbers, are shown in terms of relative risk, not absolute. They indicate the greatest probability for population growth, rather than exact definition of extinction risk (Akçakaya, 2000).

We estimated the population viability in a short (25 years) and a long (50 years) time period. Short periods are better for decisions

Table 5. Results of sensitivity analyses of population viability analysis modelling for Griffon Vulture population. Base model value for mean growth rate r was 0,103. Percentage changes in r from the *Basic scenario* are presented.

Parameter	**Changes in *r*, %; (-) means *r* is decreasing; (+) means *r* is increasing for certain percentage; 0 means that *r* remains the same**
Lethal Equivalents 3.14 -20%	-0.98
Lethal Equivalents 3.14 -10%	0
Lethal Equivalents 3.14 +10%	0
Lethal Equivalents 3.14 +20%	0
Age of first reproduction < 1 year	-9.71
% Breeding females > 10%	-7.77
% Breeding females < 10%	+6.79
Mortality age 0-1 > 10%	+3.88
Mortality age 0-1 < 10%	-2.92
Mortality age 1-2 > 10%	+1.94
Mortality age 1-2 < 10%	-2.92
Mortality age 2-3 > 10%	+2.91
Mortality age 2-3 < 10%	0
Mortality age 3-4 > 10%	-0.98
Mortality age 3-4 < 10%	-0.98
Maximum age of reproduction < 10 years	-13.59
Maximum age of reproduction > 10 years	-92.24
Population size > 10%	-1.95
Population size < 10%	0
K – carrying capacity 500	0

on population management. Consequences of our actions could become quickly evident, in order to provide feedback for further decisions. In addition, in the longer period, environment will change in a way we cannot exactly predict, which means that predictions for long time period are less reliable (Hanski, 2002). The short period models are more reliable because as the period gets longer the errors are multiplied in every step (Beissinger & Westphal, 1998).

In deterministic projections values of population growth rates r, remain the same for different time projection (25 and 50 years) since only time periods are different and other parameter values are equal. We get the highest population growth rate for the *Most optimistic scenario*, but the value of deterministic r for the *Basic scenario* is just behind. Two scenarios, *Scarce food* and *Veterinarian drug f30* have negative values of deterministic r. These results clearly show that catastrophes which have a drastic influence on the survival of individuals cause population decrease. The values of stochastic population growth are expected to be lower than deterministic values, but it is noteworthy that in the *Veterinarian drug f30* scenario, where r is negative, they are twice lower than the deterministic ones.

Analyses have shown that population viability analyses are greatly influenced by demographic or environmental stochastic factors, which is in conformity with Lande (1998). Although the *Basic model* shows a constant positive population growth rate, from the model *Massive poisoning* we could see that a long time period is needed for a small population to reach a level beyond the danger of extinction. This population size inertia has an inherent time scale of the length of time that a trend in population growth or a decline is liable to persist (Goodman, 2002). Comparing the *Basic scenario*, with the *Veterinarian drug* and *Pollutant* scenarios, we can clearly see the different influence of certain factors (in this case of veterinary drugs or pesticides) on reproduction and survival and the importance of catastrophe occurrence in projected time series. A factor which reduces survival by 50% significantly influences population decline, while a factor which reduces reproduction rate by 50%, has no significant influence on population size. A 10% frequency of catastrophe (for instance in the *Veterinary drug* scenario) also does not affect survival rates significantly, but a 30% frequency of catastrophe changes the situation drastically, and our results show that the population is threatened with extinction in a very short time.

The simulations done with the *Basic model* predict a population increase which is in accordance with the actual population growth in the past ten years on the Island of Cres.

From the results of different scenarios we could see that the highest threat to Griffon Vulture populations is increased adult mortality. It can be caused by food shortage, which consequently reduces reproduction and survival rates. The present population depends exclusively on the traditional sheep raising, since dead sheep are the main food source of Griffons.

The modelling of the population has shown that population management could be enhanced by simple protective measures and supplementary feeding, which will result in reduced mortality rates. The models *Scarce food* and *Veterinarian drug f30* have shown that the population could easily become extinct in the near future if measures are not taken to alleviate food shortage, or to stop poisoning.

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REINTRODUCTION PROGRAMMES FOR VULTURE SPECIES

David C. Houston

Institute of Biomedical and Life Sciences, University of Glasgow,
Glasgow, G12 8QQ Scotland, United Kingdom
E-mail: D.Houston@bio.gla.ac.uk

Abstract

In recent years there has been an increasing recognition of the role that reintroduction programmes can play in species conservation. Some of the best examples of successful bird reintroductions come from vulture projects. There may be fundamental features of the biology of these scavenging birds that make them particularly suitable for this management technique. IUCN has developed international recommendations on the procedures that should be followed before any species reintroduction programmes are planned. These will be reviewed in relation to their relevance for vulture projects. Fortunately there is considerable data from the reintroduction programmes for the California Condor in America and the European Griffon and Bearded Vulture in Europe, and these allow us to consider which are the most important factors to consider if reintroductions are likely to succeed. It is becoming apparent that the major challenge is probably in the selection of suitable release sites, and the management of birds once released. Vultures have extremely extensive foraging behaviour. This can expose them to a wide range of environmental hazards, and if these are not fully understood can lead to a low probability of self-supporting populations becoming established. The mortality agents acting on vulture populations are often quite unexpected, and extremely difficult to predict. Detailed monitoring of released birds is not only essential in order to identify the hazards that may face released birds, but also gives essential information for the conservation of wild populations. The role of supplementary feeding of released birds is also badly in need of more serious study. There are obvious advantages to this technique, but there may also be serious disadvantages if it prevents the development of natural normal foraging behaviour in released birds.

Keywords: Vultures; Condors; Reintroductions; Supplementary feeding.

Introduction

There is increasing acceptance that re-introduction projects can play an important role in field conservation. This approach has for some time been a main focus of the zoo community (Tudge, 1991), although many field biologists have considered it of marginal value. However in recent years there is increasing recognition that man's impact on the environment is now so profound that such interventionist forms of wildlife management will be necessary to support many endangered species. Vulture programmes have played a major part in the greater acceptance of reintroductions. Most vulture species in the wild are now undergoing population decline. Some of them catastrophically, as is the case of the three species of Indian Griffon vultures (*Gyps bengalensis, G. indicus* & *G. tenuirostris*), which within a decade have declined from being the most abundant birds of prey on the Indian subcontinent to critically endangered (Prakash *et al*. 2003). A vulture species is among the most endangered birds in the world: the California Condor at one time was reduced to only 22 individuals (Snyder & Snyder, 2000).

There are three vulture reintroduction projects which have been outstandingly successful: those for the California Condor *Gymnogyps californianus* in America, the Eurasian Griffon Vulture *Gyps fulvus* restored to the Massif Central region in France, and the Bearded Vulture *Gypaetus barbatus* back into the European Alps. What makes these projects particularly important is that all three have been the subject of major research programmes, making them among the best studied reintroduction programmes for any animal group in the world. In this short review I consider the lessons that these past projects may provide for future vulture reintroduction programmes and field management.

Role of Reintroductions in Conservation

The concept of saving endangered species by holding them in captivity, breeding, and subsequent release back into the wild goes back a surprisingly long time. Pere David's Deer *Elaphurus davidianus* was saved from extinction by the Chinese Emperors, and latterly by the Duke of Bedford in England, and is now re-established in China, having probably been extinct in the wild for 800 years. The California Condor is one of only seven such bird and mammal species which have so far been saved in this way, being totally extinct in the wild at the time of their reintroduction (the others being Arabian Oryx *Oryx leucoryx*, Przwalski Horse *Equus przewalskii*, Red Wolf *Canis rufusr*, Guam Kingfisher Halcyon cinnamominus and Guam Rail *Rallus owstoni*, Olney *et al*. 1994). Many hundred more species have been reintroduced back into parts of their range from where they had become locally extinct (Griffith *et al*. 1989). The earliest of these re-introduction programmes were not planned in any scientific way, and a major advance that has occurred in recent decades has been the realisation that the success of such programmes depends heavily on how they have been planned. Also we need to recognise that not all species are suitable: Stuart (1991) suggested only about 10% of endangered mammals were suitable for reintroduction. Vultures, however, seem one group of birds where reintroduction can be highly successful, and about a third of all the vulture species in the world are currently the subject of reintroduction projects. This suggests that there are aspects of the biology of vultures, which predispose them to successful reintroduction management.

Suitability of Vultures for Reintroduction Programmes

Both Old and New World vultures evolved as exclusive scavengers. This required them to evolve adaptations to minimise energy expenditure (Houston, 2001). The food supply available to any large scavenging animal comes largely from the carcasses of ungulates, and these are widely dispersed, transient and unpredictable in location. Such a food resource can only be exploited by a scavenger with minimal energy expenditure, because the energetic cost of reaching the food supply must always be lower than the energetic gain obtained from feeding on it. Vultures could not survive if they did not use soaring flight - which is the most energetically efficient of all forms of travel – and is the reason why vultures can so easily out-compete scavenging mammals (Ruxton & Houston, 2004). Bahat (1995) showed that another of their specialisations is the ability to lower their metabolic rate when deprived of food. This inherent characteristic of minimising energy expenditure probably predisposes vultures to settle well into captivity. Unlike wild caught falcons, owls or most other birds of prey, who when captured often exhaust themselves by flying against cage walls in an increasingly desperate attempt to escape, in my experience vultures usually accept confinement with little sign of stress. Evolution has selected passive behaviour traits in vultures, and they are temperamentally not prone to hysterical behaviour, because such actions are energetically expensive.

In the past few zoos bred vultures, largely because they made little effort to do so. There was always a sufficient supply from the wild. Virtually all species, however, have been shown to breed well in captivity when given sufficient care. In the early days of the California Condor recovery programme there were some who held the view that Condors were so vulnerable to human disturbance that they would never settle or breed in captivity. In practice, the zoos involved in this project have shown how highly skilful management can lead to impressive breeding success. There were 27 remaining Condors in 1987, when the decision was taken to bring the last of the wild birds into captivity. Within a decade of captive breeding their numbers had risen to 150 (Snyder & Snyder, 2005). This remarkable breeding success, for a bird with such an inherently low rate of reproduction, is all the more impressive because all 27 of the founder birds had by then bred, and secured their genes for the breeding programme.

Vultures also lend themselves to this form of management because of the ease of release back into the wild. It is relatively easy to assess food availability at release sites, because this is largely a function of ungulate (domestic or wild) numbers within the foraging range, and their mortality rate. Given these data, it is a simple matter to calculate the number of carcasses likely to become available each year. This is a process one cannot achieve for predatory birds of prey. They have the complication of not only needing a sufficient density of potential prey, but also that prey has to be caught and killed. Only a small proportion of predation attacks on live prey are successful, so the probability that a certain level of prey capture will be achieved by a released predator is uncertain. Furthermore, carrion feeders can have their food supply augmented in a simple way that is difficult, if not impossible, for most predatory birds. The dependence of vultures on carrion has another advantage in that there are no complex predation techniques to be learnt. Most vulture species collaborate when foraging, so if birds are released near to those already established in the wild, they will probably learn foraging skills from the behaviour of the older individuals. All of these basic features of their biology make vultures highly amenable to captive breeding and release back to the wild.

Guidelines for Reintroduction Projects

Early animal reintroductions depended on little more than the enthusiasm and whim of a dedicated patron. Today, reintroductions need a formidable amount of planning. Many countries have formal procedures that have to be followed before any species can be released into the wild. This precaution is understandable when we consider the number of introductions, often with the best of intentions, which have proved an ecological disaster. Perhaps the worst of many examples is that of the Nile Perch, which within a few decades of its introduction into Lake Victoria has already caused the extinction of perhaps half of the

300 or so species of native fish (Wilson, 1992). In an attempt to avoid such catastrophes, reintroductions, by definition, are confined to release of species back into part of their historical range, from which they became extinct. But there are still a considerable number of factors to consider if release programmes are to succeed (Kleiman *et al*. 1994). The Re-introduction Specialist Group of the IUCN produced a series of Guidelines for Re-introductions in 1995. These are not requirements, but more guidelines to be considered before contemplating a reintroduction programme. They are available on the IUCN website at http://iucn.org/themes/ssc/pubs/policy/index-1.htm. I do not intend to repeat them here, but rather to highlight some aspects that are particularly pertinent in the case of vulture reintroductions.

Source of Birds

Vulture reintroductions have used each of the three alternative sources of birds.

1) Take from the wild, usually as fledglings. Often using young birds which fall from their nests, or which are picked up in poor condition shortly after fledging and given rehabilitation. Obviously this should only be done if the loss of these individuals will not endanger the source population. Birds from nests in the Pyrenees were the source of most of the captive, and later wild released, Griffon Vultures into the Massif Central (Terrasse *et al*. 1994).
2) Captive breeding, with young birds being released at the age at which they would naturally fledge. This has been the main technique used for the Bearded Vulture reintroduction back into the Alps (Frey *et al*. 1995).
3) Captive breeding, with the birds released when they are older, either as juveniles or even adults. California Condor releases have involved birds of a wide range of ages, in California from 6 to 18 months, but with a mean of 12 months (Mee, pers. comm.).

Practical considerations will probably determine the strategy to be used in any programme, and there is little data to suggest any one method is markedly more successful than another. Sarrazzin & Legendre's (2000) analysis of the *Gyps fulvus* reintroductions in to the Massif Central showed that demographically it was better to release adults than juveniles. But a later modelling study suggested that there is a greater risk of extinction through inbreeding depression if adults are released, and for this reason juveniles are best (Robert *et al*. 2002, 2004). In planning the first of the major vulture re-introduction projects, Michel Terrasse deliberately did not initially release young birds. This was because juvenile vultures are known to disperse away from the natal area, and without an established breeding colony in the area they may not return. Instead he held birds in breeding cages at the release site for many years, and only released breeding adult pairs, in the hope that they would by then have formed an attachment to the reintroduction site. Later, as the population grew, younger birds were released, but by then there was a resident breeding population to hold the birds in the area. This was clearly a most sensible and successful strategy (Terrasse *et al*. 1994), and survival rates and breeding success of the introduced birds has been remarkably high (Sarrazin *et al*. 1994, 2000).

Genetic Stock

Ideally birds used in release programme should be from the same genetic stock as those native to the area. In practice this may be impossible to achieve. The Bearded Vultures released into the Alps came from a wide variety of captive collections, the source of some of these birds being uncertain. It is thought some originate from the Soviet Union, and Afghanistan as well as Greece and the Pyrenees (Gautschi, 2001). Over 100 birds have been released since 1986, and these derive from a long breeding programme that was initiated with 61 founder birds held in a variety of zoos and breeding centres. However only 36 of these individuals went on to actually breed. Despite this, Gautschi (2001) has shown that this captive population was genetically more diverse than the present population of bearded vultures in the Pyrenees, and more diverse than museum skins suggest the original population in the Alps and Sardinia. However, the Pyrenees population is thought to have fallen to only about 50 pairs in the 1970's and so may have then encountered a genetic bottleneck which resulted in this population having abnormally low genetic diversity. In contrast the remaining California Condors seem to show little genetic diversity compared to captive Andean Condors (Geyer *et al*. 1993) and all cluster into 3 clades which are relatively closely related. Chemnick *et al*. (2000) showed that, based on maternal DNA lines, the Condors were from limited genetic stock. Whether this will pose a problem in the future only time will tell. But hatchability of California Condor eggs at 76% is not far below what would be expected for a wild bird population, especially when you consider that many pairs were forced matings for genetic reasons, which might be expected to result in lowered breeding success (Snyder & Snyder, 2005).

One of the major advances in recent years in developing reintroduction techniques for animals has been a clearer understanding of the genetical problems that such populations can encounter. These can be summarised in three ways. Firstly there is the inevitable loss of genetic variability through holding small population sizes over time. Secondly there is the problem of potential inbreeding depression – where small population size may force birds to mate with related individuals. And finally the risk of outbreeding depression, where genetically distinct sub-populations, each adapted for different environmental conditions, may produce offspring that are less fit.

Gautschi (2001) has been able to demonstrate that there has been some loss of genetic variability during the breeding programme for the bearded vulture. This is inevitable whenever small populations of animals are held over time, but the scale of the loss can be minimised if the population is carefully managed. The aim of the genetic management of the California Condor programme is to maintain 90% of the former heterozygosity after a period of 200 years (Soulé *et al*. 1986). This is achieved by maximising the number of offspring obtained from the original founders, ensuring that all founders breed, and minimising the length of time at which populations are held at low numbers. I do not know of any evidence for inbreeding depression in vultures, although this has been well documented for some other reintroduction programmes, such as that of the Arabian Oryx. In this species the survival of calves was lowest in those mated pairs which were more closely related (Stanley Price, 1989), and similar relationships might become apparent in the future with vulture populations. Remarkably, Gautschi (2001) has been able to show that there is outbreeding depression in the captive bearded vulture population, and she found that genetically distant pairs were found to lay fewer eggs and those eggs were less likely to hatch. This is, to my knowledge, the only known demonstration of outbreeding depression in any reintroduction project.

Some have argued that the genetical risks involved in reintroduction programmes may not be all that important (Lande, 1988). Certainly for vultures I think the genetical risks involved in reintroduction programmes are low, compared to the scale of environmental problems these programmes face. This is for two reasons. Firstly vultures are birds which are known to travel widely, and so some degree of genetic exchange between isolated populations might be expected. Gautschi (2001) was able to demonstrate, through extracting DNA from a large number of museum skins, that there must have been some gene exchange between the apparently isolated former bearded vulture populations originally found in Europe, and that this was on the scale of about 1 individual per generation. Such wide-ranging species as vultures are unlikely to develop the same extent of genetical isolation in separated populations as found in species with poor dispersal abilities. Secondly, vultures are exceptionally long lived as individuals. Because genetical factors develop through generational time, not real time, any consequences may not become manifest for several decades. Even then it may be possible to correct problems by deliberate movements of individuals between populations.

Selection of Release Sites

It is becoming apparent that assessing the suitability of release sites is an extremely difficult task. The large vulture species have population characteristics similar to those of the large albatrosses. They have very low rates of reproduction, most species producing no more than one chick per year. This means that populations can only be sustained if juvenile and adult mortality are also correspondingly low. Modelling shows that only small increases in the rate of vulture mortality induced by man can cause such populations to crash dramatically (Green *et al*. 2004).

The problem with assessing risk is that the causes of vulture mortality are often totally unexpected, and impossible to predict. We can clearly identify expected threats, such as poisoning, electrocution or persecution. But it was only the intensive field monitoring of California Condors that identified lead poisoning as the major cause for their decline – the lead being derived from fragments of ammunition that the birds ate whenever they fed on deer that had been wounded or killed by hunters (Snyder & Snyder 2000). The collapse of populations of three Griffon vulture species in India has been identified as due to a veterinary drug, diclofenac. This anti-inflammatory agent, which has a long history of use in human medicine, was assumed to be relatively harmless. However, the presence of trace amounts in the carcasses of cattle is sufficient to kill any Griffon vulture which feeds on the meat (Oaks *et al*. 2004; Anderson *et al*. 2005) and this factor alone can account for the vulture population collapses (Green *et al.* 2004). The latest unforeseen problem seems to be seriously reducing breeding success in California Condors. So far captive reared birds have been released at four sites in North America: two of these in California and one each in Arizona and Baja California. But breeding success has varied between these sites. The southern California population started breeding in 2001, since when 13 eggs have been laid, 10 chicks hatched, but only one chick has fledged (Mee *et al*. in press). One of the major causes of this high chick mortality has been found to be a remarkable quantity of junk food items building up in the crops of young birds. Adults appear to routinely swallow pieces of glass, china, metal and other human debris, and pass this on to chicks with the food. Whereas there is no evidence that adult vultures have difficulty voiding these indigestible objects as pellets, it seems that the young chicks cannot always do so. All wild condor chicks examined have been found to have these junk food items in the stomach, this has directly caused the death of several chicks, and probably contributed to the weak condition of many more. This is not a problem unique to Condors: Benson *et al.* (2004) report similar problems at Cape Vulture *Gyps coprotheres* nest sites, and reports have also come from Eurasian Griffons in Israel (Bahat, pers. comm.), Pakistan (Gilbert, pers. comm.) and Armenia (McGrady, pers. comm.). This suggests the ingestion of junk, indigestible items is a common feature in many vulture species. For these reasons all potential release sites need very careful evaluation, coupled with a detailed literature review of potential risks that have been encountered by other scavenging birds throughout the world. Recent historical ranges may not be the best release sites: it seems that, because of environmental contaminants, survival and breeding success of Condors released into the relatively pristine environment of the Grand Canyon in Arizona are better than those of birds released into their last historical range in California (Snyder & Snyder, 2005).

Release Techniques

Analysis of many former reintroduction projects has identified a number of factors that are associated with success. Griffith *et al*. (1989) analysed almost 700 such projects, and among their findings was a clear relationship between the number of animals released and the probability of success. Their analysis suggests that the release of small numbers of individuals over a short period of time is most unlikely to result in a self-sustaining population becoming established. This is probably partly because small populations always have a high probability of extinction through stochastic factors, to which must be added the obvious increased risks that animals face through being introduced to a novel environment. All of the successful release programmes for vultures have released groups of birds over a number of successive years. Over 100 bearded vultures have been released into the Alps since 1986, and the Griffon vulture population was restored to the Massif Central by 61 released birds between 1980 and 1986.

Maintaining birds in captivity for long periods may cause behaviour patterns that make successful reintroduction more difficult. Hartt *et al*. (1994) showed that puppet-reared California Condors, who were then housed as groups of immatures, would later find it difficult to form pair bonds with their other cage mates. Captive birds may also not recognise risks in the environment when they are released. Pre-release training is now conducted on Condors to break them of the habit of perching on power lines. Mild electric shocks are used on experimental poles built in their aviaries to deter them from perching on artificial structures. Captive condors had also been found to be attracted to perch on buildings, and even enter houses. To prevent this, some caging for Condors is now designed to avoid the rectangular conventions of human dwellings, and instead be based on the irregular structures more typical of natural nesting caves (Snyder & Snyder, 2005). In general, management of any birds intended for release should be aimed at minimal human contact, and the use of parent rearing wherever possible.

If we accept that reintroduction projects are only likely to succeed if sufficient animals are released over a number of years, there may be a conflict with obtaining sufficient birds to achieve this with the desired genetical characteristics. Given this dilemma, it is probably best to give preference to releasing as large a number of birds as possible, even if this is at the expense of genetical considerations. This is because, as mentioned earlier, genetical problems are perhaps less likely in vulture populations than for some other animal groups.

Given the way in which vultures rely on each other to locate food sources, it is obviously desirable to release new birds close to established groups. Equally obvious is the need to monitor the released birds carefully. Not all released birds will survive, just as not all young who fledge naturally will make it to become adults. But intensive monitoring, with radio telemetry, of all released birds is often the only way of determining the threats that the birds face. Finding sick, injured or dead birds can provide invaluable information on the risks present in the environment, and this is essential for the eventual success of the project.

Post-Release Management

The use of supplementary feeding sites, usually called ‘vulture restaurants’, was first developed in southern Africa. The feeding site for Bearded Vultures at Giants’ Castle in the Drakensburgs was established in 1966. This technique was later introduced in the Pyrenees National Park in 1969 (Terrasse, 1985), following survey work that indicated all vulture species were declining. In both countries it was thought that the ‘restaurants’ would compensate for a possible decline in natural food because of changes in wild ungulate populations and farming practices in the mountains, and also avoid the risks of strychnine poisoning, which although illegal was still widely used at that time. An additional, and very important, benefit was that they acted as a focus for media interest, and educational work for residents and visitors, on vultures and their role in the whole mountain ecosystem. Other feeding stations soon followed. In the Pyrenees there was subsequently a remarkable increase in numbers of Griffon and bearded vultures in this area. The role that vulture restaurants made to this recovery is uncertain. This is because at the same time there were rapidly changing attitudes to raptor protection, and reduced persecution, together with increases in density of chamois *Rupicapra rupicapra* and other wild ungulates in the mountains (Terrasse, 1985). However, there seems little doubt that vulture restaurants made a major contribution. It is now standard practice in vulture conservation programmes throughout the world. There has not, to my knowledge, been a thorough study of the effects of these feeding stations, and I think this is long overdue. Steven Piper's paper in this volume gives a much fuller account of the costs and benefits of this feeding. The apparent benefits are often stated, but less attention has been given to possible adverse effects, and for this reason I intend to concentrate on these.

Firstly, feeding stations could prevent the development of normal foraging behaviour. If food is provided at permanent sites, then birds may become dependent on these artificial food sources and delay, or never fully develop, normal foraging skills. There is little good evidence on this. Food provisioning did not prevent California Condors from foraging widely outside the provisioned area (Wallace & Temple, 1987). But more recent telemetry work has shown that Condors released in California do not travel nearly as extensively as those released in Arizona, where food provisioning is less intensive (Mee, pers. comm.). We need much more good data on this. More attention needs to be paid to the way in which supplementary food is delivered. There needs to be an assessment of exactly what benefit is expected from the additional food, at what seasons, and what is the best method of delivery. Maybe permanent sites are not always best: if carcasses are distributed more extensively and unpredictably they may encourage the birds to develop better foraging skills, but still achieve the intended gains.

Vulture restaurants, if they are permanently established, often concentrate birds in a particular locality. This might be one of the intentions of the programme, if there are risks that the birds might face if they were to forage widely away from the protected area. But concentrating birds in predictable locations is in itself a risk. A deliberate, or accidental, poisoning incident is likely to have a far greater impact on the population if it occurs at a site where birds are concentrated in this way.

Secondly, feeding stations need to be well managed if they are to be beneficial. Some are little more than a bunch of dead farm stock dumped onto a hillside and left to rot. The source of carcasses needs to be carefully assessed and evaluated. There are obvious risks from providing carcasses that might contain high levels of pesticides. Sheep dips and other treatment for ectoparasites of stock often contain some of the most dangerous of all pesticides. The risk of drug treatment to domestic stock leaving poisonous residues in carcasses has been strikingly demonstrated by the way diclofenac has eliminated virtually all Griffon vultures from much of the Indian subcontinent (Prakash *et al*. 2003). Diclofenac, a drug which is safe in mammals, has been found to be fatal if eaten in only trace amounts by vultures. We have no information on the risks posed by the presence of other widely used veterinary drugs in dead farm-stock. Antibiotics are perhaps an area of particular concern. Apart from their use to treat infections in domestic stock, they are also widely used as growth promoters in the animal feed given in intensive units for pigs and other livestock. Dosages for growth promotion are far above those used for veterinary treatment, and in many parts of Europe carcasses from intensive pig units are the most frequently used carcasses at feeding sites. It is well known that birds differ from mammals in many aspects of the way they can metabolise antibiotics and drug compounds, and many antibiotics that are safe in mammals are definitely not safe to birds (Bishop, 2001; Altman *et al*. 1997). Surplus carcasses donated by hunters pose the obvious risk of containing lead, whose consumption will kill vultures. If old carcasses are not regularly removed, there is the risk of birds feeding on meat which is badly decomposed. There is a popular assumption that vultures can, and even enjoy, feeding on stinking meat. It is just not true. Decomposing meat usually contains a complex community of micro-organisms, most of which are harmful or which produce toxins that are dangerous to all known higher organisms. Both Old and New World vultures will, if given the choice, avoid eating such meat (Houston 1986 & pers. obs.). As birds that evolved to feed on dead carcasses, vultures do probably have more natural resistance to microbial toxins than some other animals. But I know of no experimental data on this, and I think it is a dangerous assumption to make that the quality of the food vultures consume will not have some impact on the health status of the birds eating it. Recent work on scavenging red kites has shown that the bacterial composition of the faeces of birds feeding at ‘restaurant’ sites (using mostly pig carcasses) is totally different from that of birds in areas where they are feeding on wild mammals (such as rabbits), and that some of the strains of Salmonella recovered from the faeces of birds which have been feeding on old pig carcasses are known to lead to high mortality in other birds (Blanco *et al*. in press.). There is an urgent need for more studies of this kind. And especially whether some vulture species might not face risks from consuming high levels of microbially contaminated meat. The experience with diclofenac in India suggests that bird species may differ markedly in their susceptibility to such environmental hazards.

Finally, vulture restaurants, if not well managed, also have wider effects on the environment. They not only provide food for vultures, but also for crows, ravens, rats, foxes, coyotes, jackals, hyaenas and many other scavengers. Most of these generalist scavengers are also predators, and they tend to congregate and increase in numbers in the vicinity of carcass dumps. They can impose a greater risk of predation in such areas and be a direct danger to vultures - two of the four California Condors released as adults were killed by Coyotes at feeding sites (Mee, pers. comm.). Artificially increasing the density of such predators might also impact on populations of ground nesting birds, small mammals, amphibians and reptiles and so have wider environmental consequences. Predator proof fencing around feeding sites is often used to minimise this effect, but in practice there are few fences that keep intelligent predators out for long, and this is rarely the solution that is hoped.

I am not suggesting that vulture restaurants are not beneficial. There is no doubt that they can play a major role in the success of most reintroduction projects. But there may be risks in their use as well as benefits, there needs to be more research in this area, and a rigorous assessment of likely benefits and risks should be conducted before they are used. The use of vulture restaurants should have clearly identified aims, be based on a careful evaluation of the best methods of delivering the carcasses (single, multiple or randomised sites), and strict control of the quality of carcasses supplied and the way the sites are managed so that unconsumed food is regularly removed.

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LONG TERM REINTRODUCTION PROJECTS OF GRIFFON *Gyps fulvus* AND BLACK VULTURES *Aegypius monachus* IN FRANCE

Michel Terrasse

Ligue pour la Protection des Oiseaux – BirdLife France
42 Rue Médéric – 92250 – La Garenne, France
E-mail: terrassem@club-internet.fr

Abstract

Alter 40 years of intensive conservation work to stop persecution (shooting and poisoning), to improve habitat quality, to educate the public and to ensure an adequate food supply, the situation of the four species of vultures (Griffon *Gyps fulvus*, Black *Aegypius monachus*, Egyptian *Neophron percnopterus*, and Bearded *Gypaetus barbatus*), shows an overall positive trend in France. In 1981 a reintroduction programme for Griffon Vultures began in the south Massif Central (Grands Causses). To help re-establish this species to its former breeding areas, other similar programmes started in the 1990s in the Southern Alps. The success of these programmes encouraged us to go further and prepare the reintroduction of the Black Vulture, which became extinct from these regions at the end of the 19th century. Today, the populations of Griffon Vultures in France originating from reintroductions (about 210 pairs in 2005), represent about 25% of the total for the whole country (ca. 800 pairs in 2005, i.e. more than a ten-fold increase since 1960) and this proportion is continuously increasing. Started in 1992 in the Grands Causses, and in 2004 in the Southern Alps, the reintroduction of the Black Vulture is still in an initial stage. Nevertheless, in these young populations reproduction started 4 years after the first releases, with 28 young having fledged since 1996. To help the understanding of these ambitious programmes, the following topics are analysed: releasing techniques for both species, monitoring of the released birds, monitoring of the breeding populations, feeding strategy used to help the birds to become independent from the conservation structures and avoid causes of mortality. Also some data are given concerning the movements of these new populations and the interactions with the autochthonous ones (from Spain to Greece).

Keywords: Reintroduction; Griffon Vulture *Gyps fulvus*; Black Vulture *Aegypius monachus.*

Introduction

Until the mid 20th century, the general situation of birds of prey in Europe seemed very bleak. Alter centuries of systematic persecution (for collections and pest control), we consider that only 1% of the historical raptor populations survived in the 1950s. At that time French hunters alone were known to kill (direct shooting or poisoning) more than 100.000 birds of prey per year (Bijleveld, 1974). The first important initiative to change this situation and introduce new legislation recognising the indispensable role of predation in the ecosystems, was the first Raptor Conference in Caen (Normandy) in 1964.

Very rapidly, after this date, most of the European countries recognised officially the importance of their raptor communities and the urgent need for their official protection.

In France, this came just in time. Regarding the situation only for vultures, most of the historical populations were by then extinct oron the brink of extinction.

The Griffons and the Black Vultures had been wiped out from southern France at the end of the 19th century, and only about 60 pairs of Griffons survived in the French Pyrenees in the 1960s (Fig. 1, 2) (Terrasse & Terrasse, 1967).

During the 1970s, a policy of feeding was started in order to provide food and avoid poisoning on both sides of the Pyrenees. The first "vulture restaurant" was created in Navarra in 1970 (Bijleveld, 1974), and was immediately followed by others in France.

The first wildlife films, showing the positive role of vultures, as carcass cleaners, began to have positive impact on public opinion. This new situation allowed conservationists to plan further steps towards vulture conservation and particularly the reintroduction into their historical territories.

Area and Methods

A number of review papers have been published on this topic (Bonnet *et al.* 1990; Terrasse *et al.* 1994; Sarrazin *et al.* 2000; Terrasse *et al.* 2004); thus, we shall only summarize the main parts of those works, whose details are to be found in these articles.

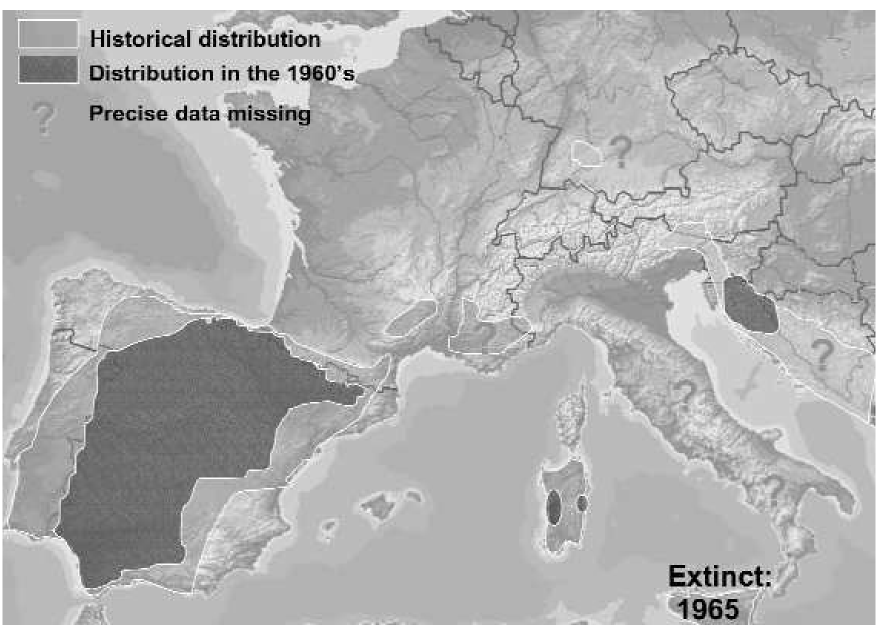

Figure 1. Historical distribution of the Griffon Vulture (19th century) in the 1960s in Western Europe.

Release Area

In 1968 the first Griffon Vulture reintroduction project began under the auspices of the "Fonds d'Intervention pour les Rapaces" and with the help of the National Park of the Cevennes, in the Grands Causses area, in the south Massif Central. The success of this programme enabled us to build up a restoration strategy for both Griffons and Black Vultures for the whole of southern France.

The study area covered all the Grands Causses and adjacent canyons of the southern Massif Central and the Provenηales Pre-Alps (Fig. 3). These areas contain cliffs and limestone mountains or plateaus, where important sheep rearing is still present (Terrasse *et al.* 2004), a good vulture habitat.

Reintroduction Protocols

For the Griffon Vulture reintroduction we have used a particular technique which is to release first adult birds (more than 4 years old) and after they had established themselves in a stable colony, to release immature birds. All of them have been kept in captivity (one or two years at least for the first adults and some months for the immature birds), in an aviary built at the release site with a view of the future breeding cliffs.

From 1981 to 1987, 61 Griffon Vultures have been released in the Jonte Gorges (northern Grands Causses) and 50 in Navacelles (southern Grands Causses).

Following the success of this programme, three new releasing places were chosen in the Mediterranean Alps (Diois, Baronnies and Verdon Canyon), where 210 birds have been released from 1996 to 2005.

At the beginning, the released stock was of different origins: Born in zoos or breeding centres, seized from poachers, recovered in the wild, etc. From the 1980s it became easier to get Griffon Vultures from Spain and later from the Pyrenees, most of them being captured, exhausted or unable to fly during the first months of their life.

Two techniques were successfully used for Black Vulture reintroductions from 1992 to 2005. The "hacking" technique consists of releasing nestlings only, from an artificial nest

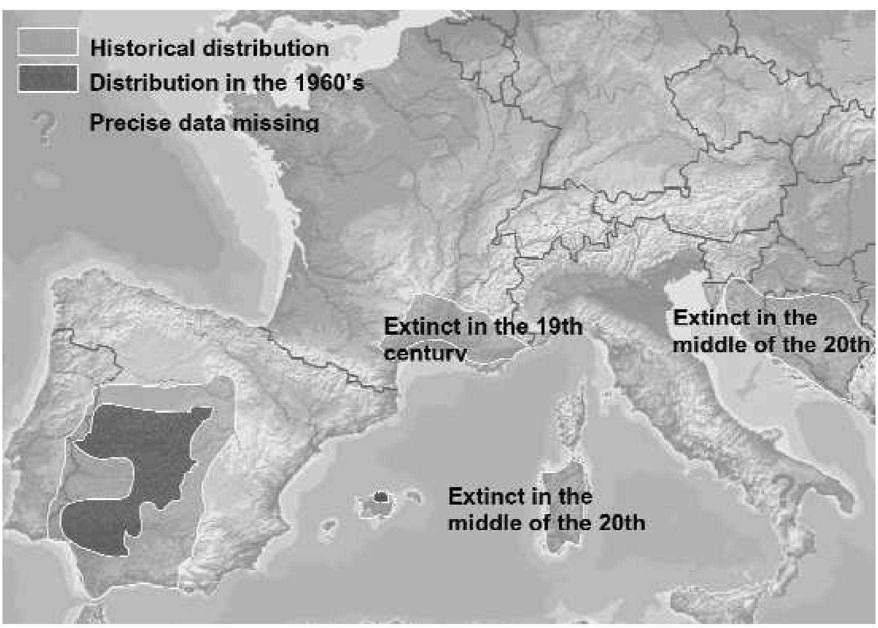

Figure 2. Historical distribution of the Black Vulture (19th century) in the 1960s in Western Europe.

site very similar to a wild nest. Thirty six birds released by hacking were born in captivity and originated from zoos or breeding centres in Austria, Belgium, France, Germany, Italy, Spain and Switzerland.

The other technique called "cage" technique, consists of releasing juveniles or immature birds, kept in captivity in an acclimatization aviary built in front of the vulture colony. 34 Black Vultures released with this method, came from rehabilitation centres and National or Natural Parks in Spain (Andalucia, Extremadura, Valencia, Castilla, Murcia).

Public Education

The first project which was launched in the Grands Causses in 1968, had to suspend release of the first vultures until 1981 because of poisoning threats. That delay was efficiently exploited to organize an accomplished education programme in the entire region in order to prepare local attitudes for the return of the vultures. At first, this activity was addressed to hunters with leaflets, posters, cartoons; it later addressed farmers and livestock-keepers by means of countless lectures and film presentations even in the smallest farms of the region. Later, with the presence of increasing populations of vultures foraging farther in areas where they were unknown, a special information letter was produced to help local people become accustomed to these scavengers.

Complementary Feeding

Taking into account that the shepherd community had got out of the habit of living with vultures, to promote complementary feeding has probably been the more important task to fulfil. Two types of feeding places were used. The first were supplied right from the beginning of the programme with carcasses mainly coming from farms and sometimes slaughterhouses.

In 1998 the veterinary regulations changed, recognising the role of vultures as natural collaborators in the sanitary disposal of carcasses. New feeding places (called *farmer feeding places*) managed by the shepherds themselves, but under our control, could then be set up (Terrasse *et al.* 2004). In 2005, the access to food for the vulture colony in the Grands Causses is as follows:

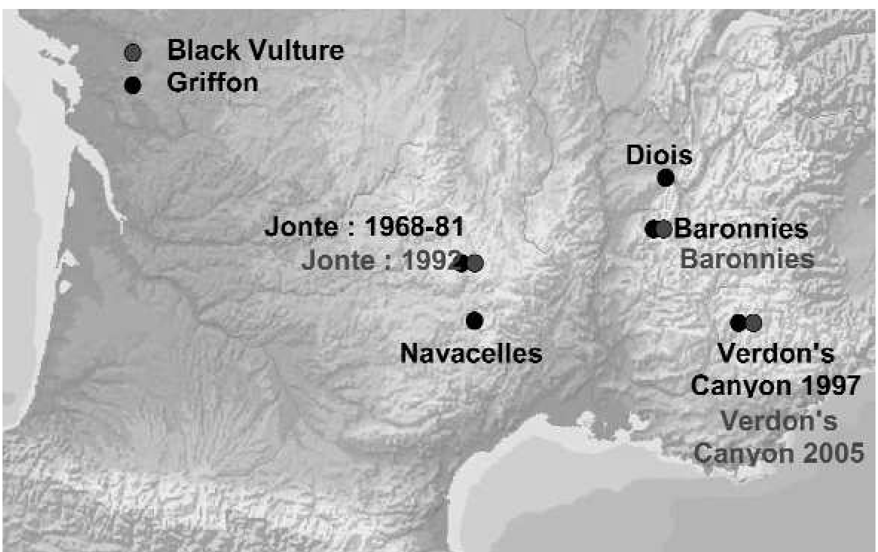

Figure 3. Locations of Griffon and Black Vultures reintroduction places in Southern France and year of first releases.

- 5 feeding places are run by the LPO (Ligue pour la Protection des Oiseaux) and the National Park of the Cevennes and are supplied with carcasses mainly coming from farms.
- 16 farmer feeding places are supplied directly by the shepherds.

Monitoring

Without describing in detail the monitoring techniques, it is important to point out that the vultures in these colonies, have, right from the beginning of the reintroduction, been continually identified and monitored. All the released birds, and most of the wild born Griffon or Black Vultures, have been ringed (metallic or coloured plastic with letters). In addition, the cage-released birds were fitted with radio transmitters, the Black Vultures being also marked with the bleaching of some wing or tail feathers.

Some birds lost their plastic rings, and to allow the monitoring to continue recapture sessions were organised. Regular re-sightings of all birds at close distance, particularly when vultures were feeding, have been very productive. In this way, 1,375 Griffon Vultures were ringed and identified and about 30,000 identifications of Griffons, and more than 3,000 of Black Vultures, were collected in the Causses colony alone. The same monitoring has been conducted in the Alps.

Results

Post Release Adaptation to the Wild

Griffon Vulture**:** This important part of our conservation work, has been tackled with conservation and demographic approaches (Bonnet *et al.* 1990; Sarrazin & Legendre, 2000). In general, the releasing of adult birds has been successful, despite some difficulties in adaptating to the wild. The annual survival rates for released adults were significantly lower for the first year following the release, but extremely high thereafter, whereas birds released as juveniles or immatures had a lower survival rate in the first two years after release (Sarrazin *et al.* 1994).

In the Causses, out of 59 released adult Griffon Vultures, 8 have been found dead and 5 were brought back into captivity. This percentage of loss (about 20 %) has been more or less the same in the Alpine programmes.

Black Vulture**:** The first reintroduced Black Vultures were juveniles (hacking), and were released in the neighbourhood of a Griffon Vulture colony. Their flying abilities were much better than the Griffons, and they immediately adapted themselves to the feeding places. The Scotch Pines *Pinus sylvestris* near the hacking site, were also used as roosting places. No bird older than two years was released (from the cages). The overall losses were not significantly lower than those recorded for Griffon Vultures.

Reproduction and Increase of the Colonies

Griffon Vulture: The release of adults have been followed, in all the reintroduction programmes (Causses and Alps), by successful breeding in the nearby cliffs within some months (Causses) or less than two years (Alps).

Since 1982, in the Causses, the breeding population gradually increased, reaching about 140 pairs in 2005. 675 fledglings have been produced in this colony within 24 years. The breeding success has steadily improved and stabilized around 0.7 young per nest per year (Fig. 4). In the Alps the result is similar, with a more rapid increase of the breeding population, mostly caused by a larger number of released birds. With 70 pairs in 2005, only 8 years after the first breeding attempt, and 127 fledglings produced within this period, these young colonies show a very encouraging progress, despite the few contacts between them. It is very difficult, however, to accurately estimate the total population size of these colonies. Indeed, this basically depends on local survival and birth rates, as well as immigration and emigration. A crude, though optimistic, count of released birds, fledglings, identified immigrants, dead recoveries and definitive recaptures, gives an estimate of maximum population size. This number is more than 800 individuals in the Causses colony at the end of 2005. It could be revised more realistically to around 600 birds. Precise censuses made in the Alps have resulted in 300 Griffon Vultures in the autumn 2005, and 70 breeding pairs.

Black Vulture: Four years after the first release, a successful breeding occurred in 1996. From this first breeding to 2005, 68 breeding attempts have been recorded, producing 28 fledglings (Fig. 5). The relatively low breeding success might be linked with the immaturity of the birds. Indeed some birds began the breeding cycle at two years old, and in 2003 two pairs successfully bred although one mate was only three years old. All the nests have been built on Scotch Pines. Being both inexperienced and without old nests, the pairs had to face different problems, linked

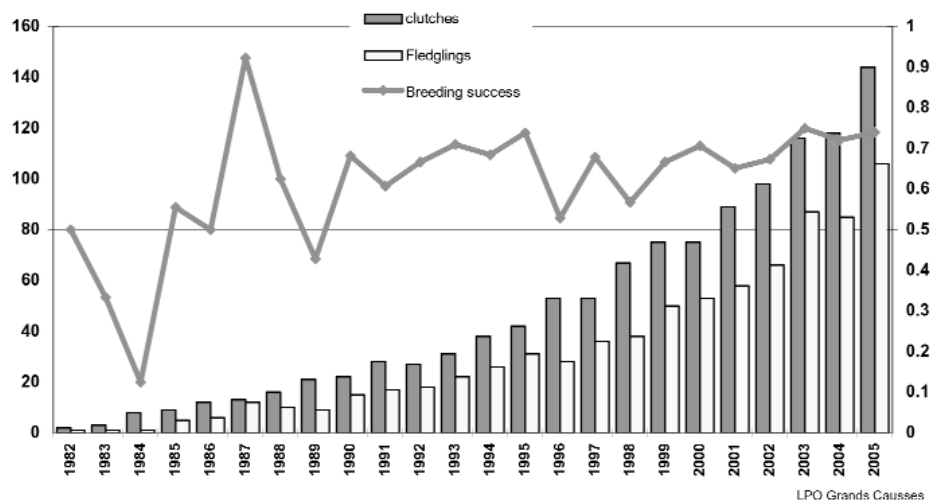

Figure 4. Number of breeding events, fledglings and breeding success of Griffon Vultures since the first releases: a) in the Grands Causses, b) in the Southern Alps.

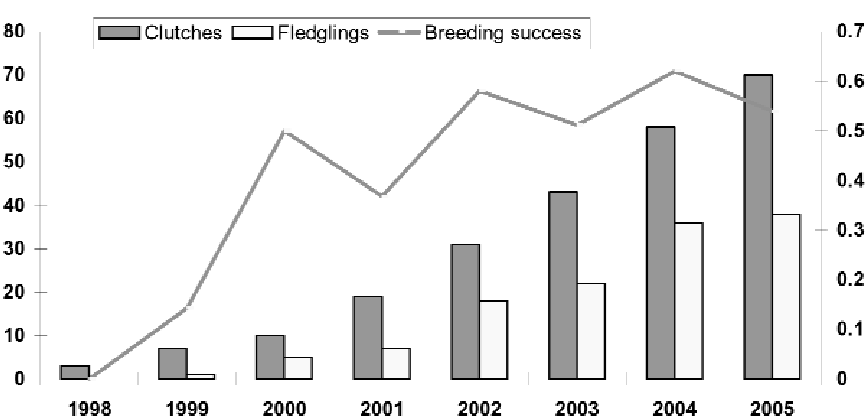

Figure 5. Breeding parameters of the Black Vulture population reintroduced in the Grands Causses.

with bad weather conditions: gales, snow or hail, which could destroy the nest or kill the young. As expected, the number of failures at the fledging period has been abnormally high. At the end of 2005, the size of the breeding territory of the 17 pairs was about 4,000 km^2, along the wooded slopes of the Jonte and Tarn Gorges.

Mortality Causes

Any reintroduced population requires a careful monitoring of the mortality causes in order to find ways to eliminate non-natural ones. In spite of veterinary studies (Philippe, 2001) the cause of 38% (Griffon Vultures) and 30% (Black Vultures) of deaths remained unidentified, since dead birds were usually discovered too late. In total, 231 Griffon Vultures were found dead in 26 years in the Causses and the Alps (Table 1).

19% of losses could be attributed to inexperience (mainly juvenile birds) and this proportion would be higher if unidentified causes could be clarified.

Among the anthropogenic causes, the high frequency of electrocution and collision (about 31% of the total) lead us to consider the protection of the birds from electric lines as a priority for every programme. In contrast we can emphasize the few cases of deaths due to direct persecution (only one case of shooting in Senegal and 6 poisoned birds).

Eighteen (18) Black Vultures were found dead between 1992 and 2005 (Table 2). In general there are fewer data than for the Griffon, but it seems that this species is less sensitive to the main anthropogenic cause of death: electrocution and collision (17%). Only one bird was poisoned in a garbage dump and two birds were shot (one of them while consuming poultry in a farm)

Among the accidents, two anecdotal events

Table 1. Mortality causes (%) of Griffon Vultures *Gyps fulvus* released or/and born in the Grands Causses and the Alps (n = 231, 1980-2005).

Unidentified	38
Electrocution	23
Starvation, Inexperience	19
Wire collision	8
Accident	6
Poisoning	3
Injury, old age	1
Imprinting (???)	1
Shooting	<1
Disturbance (helicopter)	<1
TOTAL	100

Table 2. Mortality causes (%) of Black Vultures *Aegypius monachus* released or/and born in the Grands Causses and the Alps (n = 18, 1992-2005).

Unidentified	32
Inexperience	17
Electrocution	16
Persecution	11
Poisoning	6
Injury (attack be ostrich)	6
Accident	6
Collision with train	6
TOTAL	100

must be pointed out: one bird was killed by a tiger *Panthera tigris* in the Salzburg Zoo, and another bird was apparently attacked and injured by a male Ostrich *Struthio camelus* in an ostrich farm in Spain.

Foraging Strategy

One of the main objectives of these programmes, after the establishment of colonies, has been to facilitate the spatial expansion and self-sufficiency in feeding of the breeding groups.

In the Causses area, we can follow the progress of the foraging behaviour for about 25 years. Until 1984, the foraging area was small (about 200 km^2) and restricted to around the release site (Fig. 6).

With younger birds joining the population, and a new feeding place in Mıjean to the west of Causses, we observed an increase of the foraging area to about 600 km^2 in 1991. Random feedings within this range were the result of better experience in terms of flight performance and knowledge of flight conditions. The releases of vultures in the south of the Causses (Navacelles) were immediately followed by regular movements to the south of this region.

Finally, after 1998 the new feeding strategy, based on farmer feeding places, has been the best tool to attract vultures to new places, with a clear increase of their foraging area. With about 4,000 km^2 visited daily by the birds (except in winter) in 2005, we have now succeeded in having a Griffon Vulture colony, increasingly independent of complementary feeding (Fig. 6).

Movements and Dispersal

Griffon Vulture: The movements of Griffon Vultures far away from their breeding or foraging territories, which was exceptional in France until the end of the 70's, increased regularly after the first releases in the Jonte Gorges, and more rapidly when the southern Causses had been visited after 1994.

But the trend increased more obviously after 1996, with the presence of Griffon Vultures in the southern Alps (Terrasse *et al.* 1994; Terrasse *et al.* 2004).

There is ongoing research about this new phenomenon. In summary, two kinds of movements are currently observed in France and in Western Europe.

- In spring, and even more during the summer months, clear dispersal movements

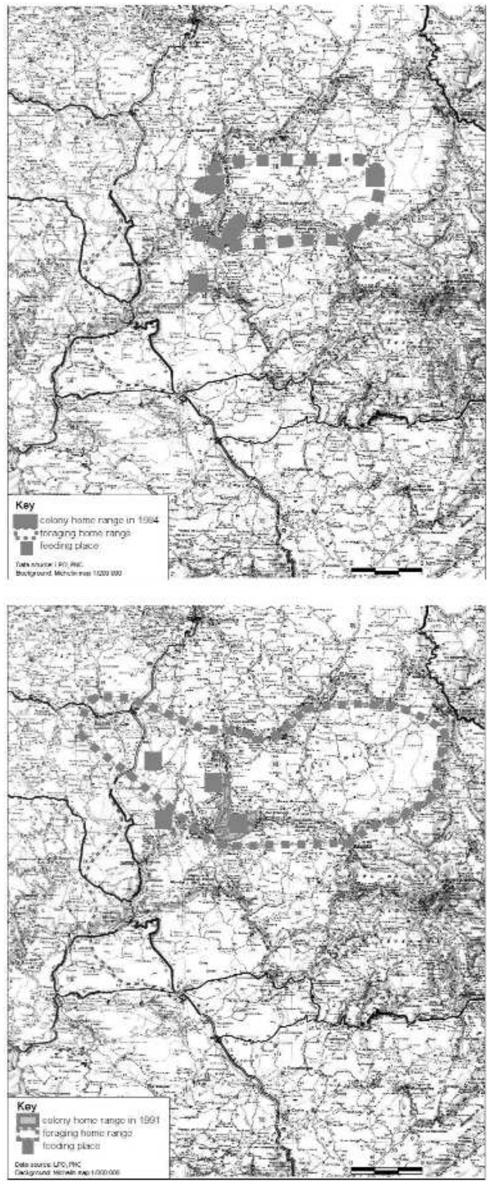

Figure 6. Foraging area in the Grands Causses in 1984, 1991 and 2004.

lead Iberian and Pyrenean Griffon Vultures to the southern Massif Central, and from the Causses colony to the Southern Alps. Some of the birds stay there (a Spanish Griffon Vulture is paired in the Causses colony). But most of these juveniles or immature Griffons, after following the improving weather to the north, can reach Switzerland, Germany, Holland, or even further away.

A similar seasonal movement leads birds from Croatia to Italy, and these vultures arriving in the Verdon Canyon can reach the Causses and return.

Recently (probably following this route), a Griffon Vulture born and ringed in the Causses in 2004, was observed in the Dadia colony in Greece, in September 2005. This bird was observed again in his native colony in November.

- In autumn, following the normal south-westwards movements, the juveniles from the Alps and the Causses, reach the eastern Pyrenees and join the Iberian birds till the south of Spain, or even Senegal.

Black Vulture: Since the beginning of the reintroduction project in 1992, large movements of Black Vultures were observed in France. More or less similar to those of the Griffons, birds from the eastern Pyrenees were mainly observed in spring and summer at the Massif Central, the Alps and sometimes as far away as Switzerland, Germany or Holland.

Some of them have shown the ability to travel huge distances, following the Mediterranean shores of Spain up to Gibraltar and returning one year later. One Spanish bird, released near Madrid, joined the French colony in the Causses, where he is now breeding. Recent data concerning the new Alpine releases show another route, leading birds to Italy and down to Sicily. One of them crossed the Mediterranean to reach Corsica and Sardinia.

Conclusions

After 37 years of intensive work, we can argue that all parameters corroborate the success of this long-term enterprise. With at least 210 breeding pairs (about 25 % of the French population) of Griffon Vultures and 17 pairs of Black Vultures, these new populations are now well integrated in these regions in what is now called a sustainable reintroduction. The scavenging habit of the vultures is well accepted and recognised through the new feeding regulations. In addition, vultures are used by local politicians to advertise Green Tourism, since they are able to attract a new public of birdwatchers from other countries during the “low” tourist season.

Among the positive roles played by these new populations, is the attraction of other species. At least one new pair of Egyptian Vulture *Neophron percnopterus*, is regularly observed around every new Griffon colony, helping this threatened species to re-establish its former breeding grounds. The increase and the resumption of ancient movements is the last positive consequence, directly linked with these new populations. Each new reintroduction programme seems to speed up the extension of the movements, acting as a relay.

In general, these movements form a link between the healthy Pyrenean or Iberian populations and the still threatened Croatian or Italian colonies. They are not only a positive factor for the restocking of Balkan populations, but also a symbol of hope for the future.

Acknowledgements

This big conservation achievement, result of half a century of work, has been made possible through a huge investment at a national and international level. A great number of individuals, institutions, regional and national governments, parks, reserves, zoos, rehabilitation centres, etc. have played an essential role in making possible the return of the vultures in France.

Among them I would like especially to thank those who believed that this idea was not a

utopian dream: Maarten Bijleveld, Jesús Elosegui, Hans Frey, Jesús Garzon, Rafael Herredia, Evelyn Tewes, Juan Sanchez. The Black Vulture Conservation Foundation and the Frankfurt Zoological Society played a major role.

I cannot mention the countless people who worked in France during these years in the Fonds d'Intervention pour les Rapaces, the Ligue pour la Protection des Oiseaux, Vautours en Baronnies, Vautours en Haute-Provence, the National Park of the Cevennes, the Natural Regional Park of the Vercors, the Museum National d'Histoire Naturelle, etc.

But I cannot forget the local people and shepherds who trusted us when it all began. Without them nothing would have been possible.

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STATUS OF THE EURASIAN GRIFFON VULTURE *Gyps fulvus* IN ITALY IN 2005

Fulvio Genero

Friuli Venezia Giulia Autonomous Region
Lago di Cornino Nature Reserve
6, March 23rd str., I-33100 Udine, Italy
E-mail: genero@tin.it

Abstract

In Italy, in recent years the Griffon Vulture has survived only in a part of Sardinia along with summering birds in the eastern Alps. The difficult situation in which the species found itself required the launch of conservation projects in Sardinia and releases in the Alps, Apennines, Sicily and Calabria. In Sardinia 60 Griffons were released in total, to halt the decline of the population. The first releases in the eastern Alps took place at the late '80s whereas the release of approximately 60 birds helped create a breeding colony that in summer attracts good numbers of birds from Croatia and other countries, totalling 60-110 birds.

In the Apennines the release of birds started in 1994, with a total of 97 Griffons released. This colony is doing especially well and the population has grown to 150-160 birds. In Sicily, two projects, started in 1998, have helped create a small colony that started breeding in 2005. In 2004 releases began in the Pollino National Park where there are roughly ten free-flying birds. In 2005 the total Italian population stood at 320-390 birds with 73-86 stable pairs formed.

The realisation of these projects has guaranteed the survival of the Griffon Vulture in five different areas of Italy and allowed a greater exchange of birds. At all of these, supplementary feeding areas were found to be indispensable. The most serious threats are the occasional use of poisoned baits, various forms of disturbance, organisational and management inadequacies when dealing with long-tem problems and the installation of new power-lines and wind-power generation projects.

Keywords: Eurasian Griffon Vulture; Conservation; Italy; Population decline; Supplementary feeding.

Introduction

The Griffon Vulture *Gyps fulvus* is extinct in most of Italy (Genero, 1992). From 1900 onwards the only nesting population was confined to Sardinia and roughly 100 - 150 individuals frequented the eastern Alps during the summer months. In the past its distribution was much wider and covered almost all mountainous areas of Sicily and Sardinia. It was said to nest in Alpine areas until the end of the 18th and the beginning of the 19th century while its presence in the Apennines is testified to in historic documents and seems probable on the basis of the species' ecological and faunistic requirements - but not beyond the 17th century (Genero, 1992). The general decline of the species can be attributed to human persecution, the use of poison baits, the modernisation of stock-rearing and veterinary practices, habitat loss and fragmentation (Cramp & Simmons, 1980). There are six ongoing conservation projects in Italy (Avesani, 2004).

Conservation and Reintroduction Projects in Italy

In Sardinia (Helmar Schenk and Mauro Aresu, pers. comm.) the Griffon Vulture was still well distributed in 1945 with a population of between 1,000 and 1,400 individuals but a rapid decline reduced the population to 65-75 individuals with 20 nesting pairs at the start of the 1980s, present in the north-western sector of the island. Numerous conservation efforts were carried out by World Wide Fund for Nature, (WWF-International), Lega Italiana Protezione Uccelli - BirdLife (LIPU), Fonds d'Intervention pour les Rapaces (FIR), Legambiente Sardinia, the Sardinian Regional Government and the municipality of Bosa, with programmes of public awareness, surveillance, supplementary feeding and releases. Between 1987 and 1995 sixty Griffons were released (Table 1), raising the population to 42 pairs in 1996 but some episodes of poisoning again reduced the population to 23 pairs in 1999. The population is now slowly

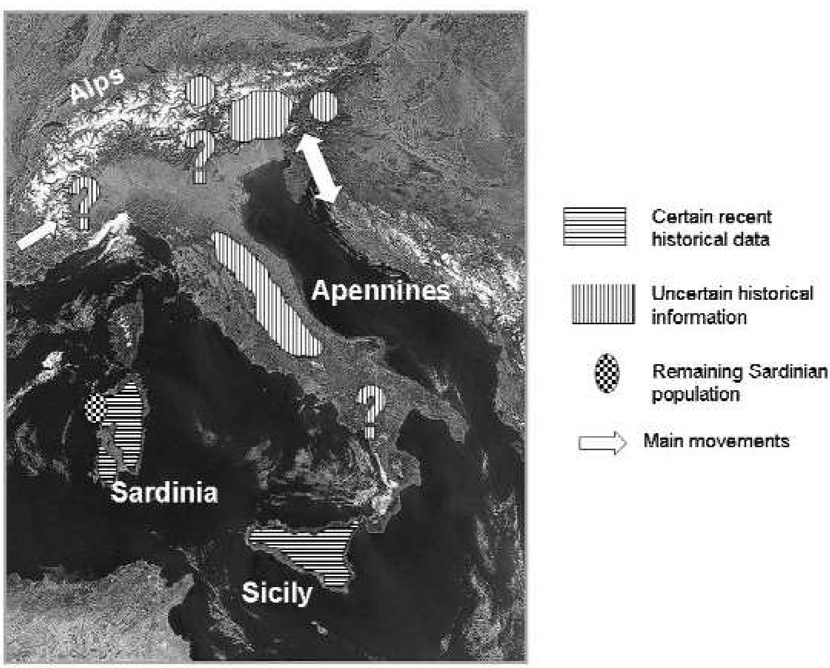

Figure 1. Historical distribution of the Eurasian Griffon Vulture in Italy.

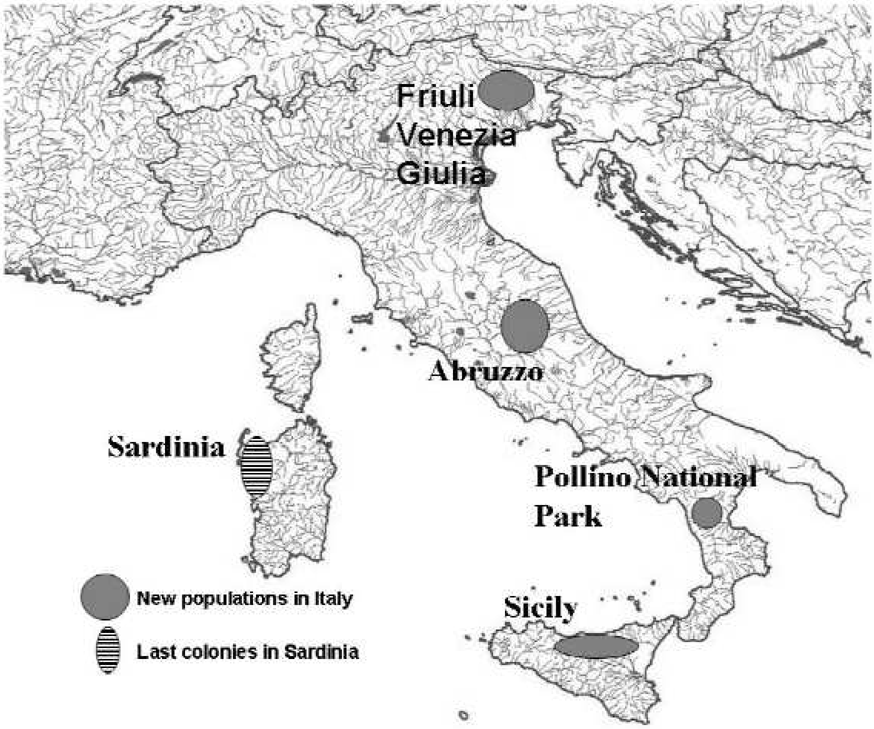

Figure 2. The distribution of Eurasian Griffon Vulture colonies in Italy.

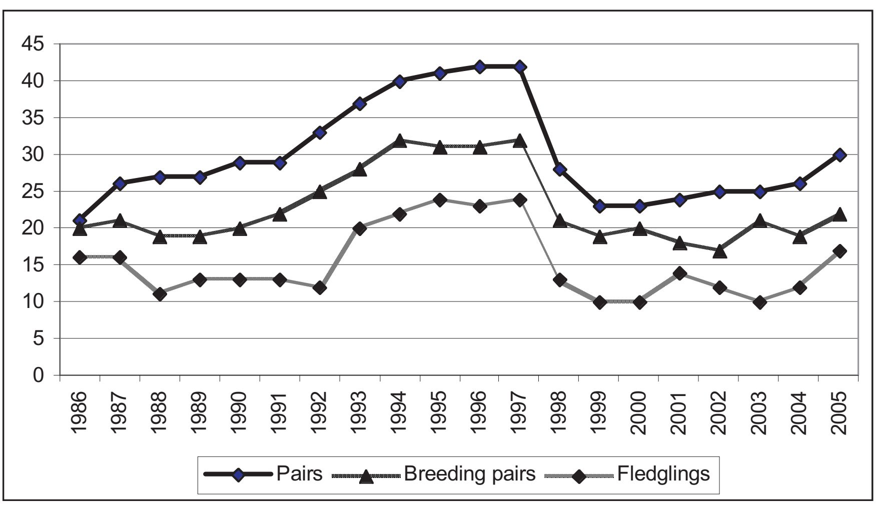

Figure 3. Trends in numbers of breeding Eurasian Griffon Vultures in northeastern Sardinia, 1987 – 2005 (from Aresu & Schenk, 2005 and unpublished data).

recovering with 29-31 pairs and 85-90 individuals present in 2005. Human disturbance is still the main cause of the failure of breeding attempts and the loss of chicks (Schenk *et al.* 2005).

In the eastern Alps a conservation project was started at the end of the 1980s with the aim of consolidating the species' presence in the Alps and creating nesting colonies (Genero & Perco, 2003). Between 1992 and 1999 a total of 60 individuals were released (Table 1). The colony exercised considerable attraction to birds summering in the Alps and their numbers have since increased to reach a level of 50 to 60 individuals at any time during the summer. These Griffons arrive largely from Croatia but also from other countries (France, Greece, Bulgaria Spain, Israel and Austria, pers. obs.). The first individuals turn up in March or April but the great majority arrive in the second half of May and June. Other arrivals are recorded at the end of August or during September when some young birds of the year appear, together with a few adults that have finished the breeding cycle. From the end of September the presence of wild birds begins to fall off with the majority of departures taking place during the month of October. For the last few years an increasing number of birds (about 15 in recent years) have begun to winter in the area joining the colony on a permanent basis. In conclusion the total number of birds in the area is 60-110. The current breeding population is roughly 15 pairs. In 1993 the first nesting attempts were recorded. Since then the number of pairs has increased but nesting success has been low, with a total of 20 birds fledged to date. The main causes of low productivity are thought to be disturbance from military aircraft and paragliders and also predation by Ravens *Corvus corax* which are very abundant on the cliffs close to the supplementary feeding site.

The colony and the supplementary feeding point attract other raptors including some species which are locally uncommon. Every year Red Kites *Milvus milvus* are recorded while since 1992 there have been eight proven records of Egyptian Vulture *Neophron percnopterus*, four of White-tailed Eagle

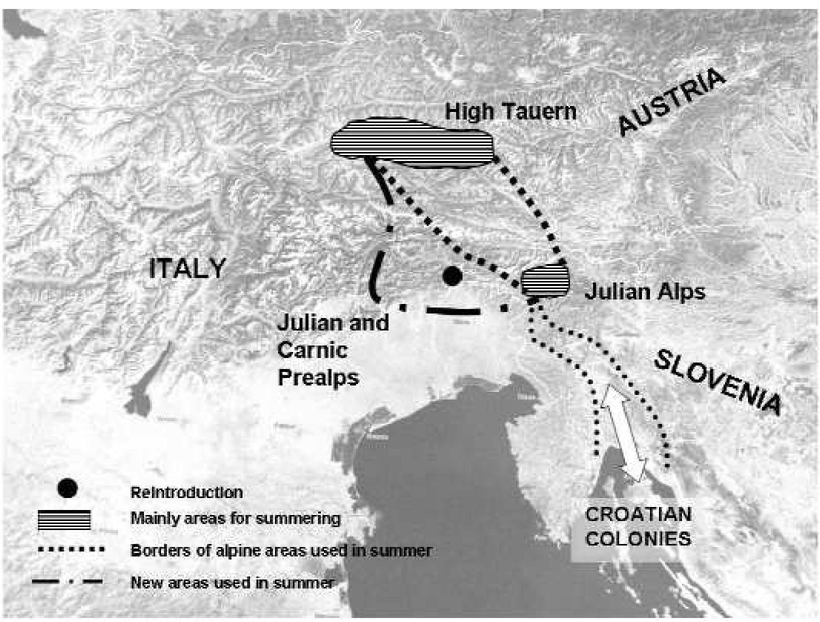

Figure 4. Presence of Eurasian Griffon Vultures and the reintroduction project in the eastern Alps.

Haliaeetus albicilla, one of Lesser Spotted Eagle *Aquila pomarina*, one of Imperial Eagle *Aquila heliaca* and one Black Vulture *Aegypius monachus* which had been released at the Baronnies in France (pers. obs.).

The project is financed by the autonomous Region of Friuli Venezia Giulia and managed by the municipality of Forgaria nel Friuli.

The reintroduction of Eurasian Griffon Vultures to the Apennines (the Nature Reserve of Mount Velino) began in 1994 and is managed by the National Forest Service (Stefano Allavena, pers. comm.). From 1994 to 2002, 97 Griffons were released in two areas 50 km apart (Table 1). Thanks to the high suitability of the area, the population has enjoyed a rapid increase with the presence of 25-35 pairs and the current population is 150-160 individuals. There are three nesting areas, 5 to 15 km apart. The first breeding attempt (4 pairs) took place in 1997. Since then the number of breeding pairs has increased constantly and more than 20 nestlings were fledged in 2003. The area frequented by the birds is quite large and includes various national parks and protected areas (about 54,000 ha). Mortality has always been very low except in 1998 when eight birds died because they fed from the carcass of a poisoned horse. Increased mortality from collision with power-lines and the possible construction of wind farms (Allavena & Panella, 1999; Allavena, 2003) are also threats to the colony.

In Sicily the species was common until the mid 1900s (Iapichino & Massa, 1989) and the last breeding population survived in the Nebrodi Regional Park until 1965 (Priolo, 1967), when the species became extinct due to a poisoning episode. Reintroduction began in 1998 with the help of LIPU, with the first releases in 2000 in the Regional Park of the Madonie and in 2001 in the Nebrodi Regional Park, situated 60 km away. About 35-40 individuals were involved (Table 1). After a few months all the Griffons moved to Nebrodi, with quite a high level of losses probably linked to the lack of the supplementary feeding sites and collisions with medium- and high-voltage power lines. The current population is 16-20 individuals. Two pairs bred suc-

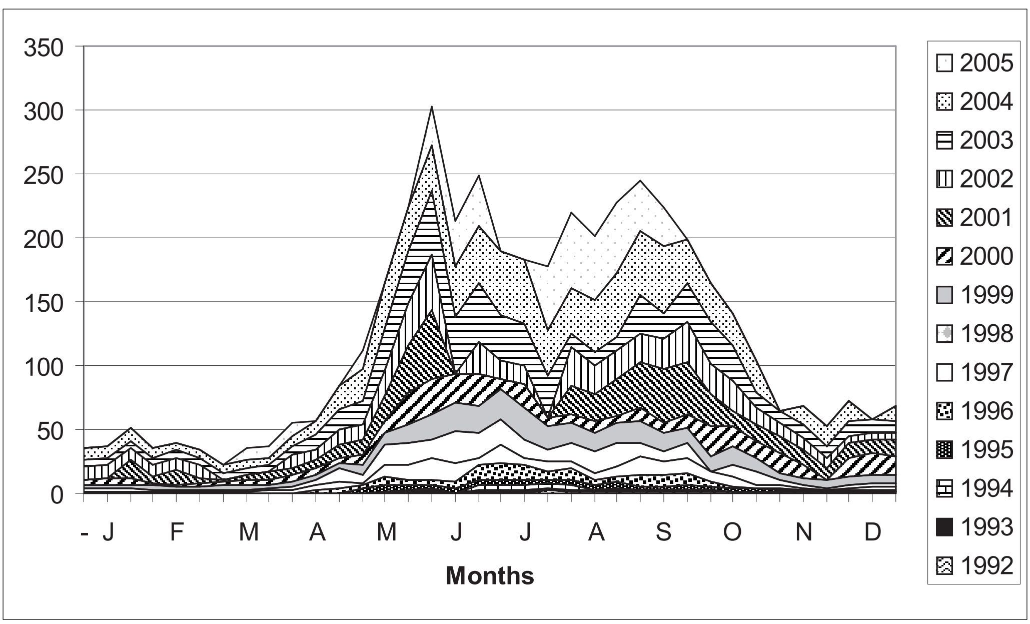

Figure 5. Numbers of non-native Eurasian Griffon Vultures recorded in each ten-day period in the Eastern Alps for the period 1992-2005.

cessfully in 2005. The release of further Griffons is planned and two supplementary feeding sites have recently started operating. The project is currently managed directly by the Nebrodi Regional Park (Massimiliano Di Vittorio, pers. comm.).

In the Pollino National Park 12 Griffons were released in 2004 (Table 1) and 9 individuals are currently present. In 2005 a single pair nested and laid an egg. The area enjoys good environmental quality and feeding possibilities (Massimo Pandolfi, pers. comm.). Further releases are planned for the future. The project is being carried out by the Pollino National Park and the Zoology Laboratory of the University of Urbino under Massimo Pandolfi.

Table 1. Numbers of Eurasian Griffon Vultures released at the various sites in Italy.

Years	Sardinia	Eastern Alps	Apennines	Sicily	Pollino N.P.	Total numbers
1987	20					20
1988						
1989	28					28
1990						
1991						
1992		12				12
1993		7				7
1994		3	6			9
1995	12	6	15			33
1996		7	10			17
1997		7	28			35
1998		11				11
1999		4	2			6
2000				7		7
2001			17	22		39
2002			16+3			19
2003		1				1
2004					12	12
2005		2		6		8
TOTAL	**60**	**60**	**64+33=97**	**35**	**12**	**264**

Evaluation and Discussion

The conservation projects carried out in Italy have permitted the continued survival of the last native population of Eurasian Griffon Vultures in Sardinia as well as the creation of new colonies in four different geographical areas. Two projects (eastern Alps and the Appenines - Abruzzo) can be considered ‘finished’ in as much as no further releases of birds are foreseen, whilst in Sicily, the Pollino National Park, and perhaps Sardinia, releases will continue for a number of years. There are various proposals for new projects that have not yet been placed in the context of the global conservation strategy for the species. Priority would appear to be for future projects in Sardinia aimed at favouring the decolonisation of the entire island and the return of both Cinereus (Black) *Aegypius monachus* and Bearded *Gypaetus barbatus* Vultures as well as the pre-Alpine area with a view to creating further points connecting the populations of eastern and western Europe.

In Italy, between 1997 and 2005, a total of 264 Griffons were released (Table 1), the vast majority being of Spanish origin. The current situation is a population of 320-389 Eurasian Griffon Vultures with perhaps 73-86 breeding pairs.

For the various vulture conservation projects in Italy there are a number of common problems. Important among these are sustaining the long-term efforts and securing adequate funding. Some projects are managed by bodies or local authorities that do not always have the capacity and the means necessary to carry out long-term and difficult projects. Other problems are linked to the use of poisoned baits and although they have not thus far caused high mortality levels, are sill employed in various parts of the country. As far as direct persecution is concerned there are only few cases of illegal killings in recent years. Disturbance of breeding birds is a problem in some areas (Sardinia and eastern Alps) and is connected to the presence of photographers, activities linked to forestry or hunting, as well as flights by helicopters, aeroplanes and paragliders near the nest-sites and feeding areas. Losses to collisions with cables (powerlines) and electrocution remain at a low level but planned wind turbine projects represent a worrying development.

The availability of food is still high in Sardinia (Schenk *et al.* 2005) where, however, the provision of supplementary feeding sites is still believed important in maintaining populations. Stocks of domestic ungulates are still high in Sicily (Di Vittorio, pers. comm.), where, though, two new supplementary feed-

Table 2. Population sizes, breeding pairs, number of supplementary feeding places and prospect of future releases of Eurasian Griffon Vultures at the various colonies in 2005, in Italy.

Colony	Number of birds	Pairs	Feeding places	Future releases
Sardinia	85-90	29-31	2	?
Eastern Alps	60-110	15-16	1	No
Apennines	150-160	25-35	1	No
Sicily	16-20	3	2	+
Pollino N.P.	9	1	2	+
Total	**320-389**	**73-86**	**8**	

ing sites have recently been created following heavy earlier losses. In the Apennines, estimates of the percentage of wild food found by the birds, range from 30-50%, according to the season (Allavena, pers. comm.). In the Alps the winter period is particularly difficult but summer grazing activity does provide some food. The increase in wild ungulate populations linked to better hunting management practices could increase food availability throughout the year.

For the management of feeding sites, efforts are made to obtain free slaughterhouse wastes and carcasses unfit for human consumption. In Friuli-Venezia-Giulia agreements with the health and forestry authorities permit the use of road-kills [especially Roe Deer *Capreolus capreolus*, Wild Boar *Sus scrofa* and Red Deer *Cervus elaphus*; (pers. obs.)]. It is hoped to establish other supplementary feeding points (but see EU Regulation 1774/2002), especially when one takes into account the high availability of ungulate carcasses that are in any case disposed of in other, often very costly ways (pers. obs).

Acknowledgements

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GENETICS OF RESTORED POPULATIONS OF GRIFFON VULTURE *Gyps fulvus* IN EUROPE AND IN FRANCE

Pascaline Le Gouar[1*], Fabienne Rigal[1], Marie-Catherine Boisselier-Dubayle[2], Sarah Samadi[2], Christian Arthur[3], Jean-Pierre Choisy[4], Ohad Hatzofe[5], Sylvain Henriquet[6], Philippe Lécuyer[7], Christian Tessier[8], Goran Susic[9] & François Sarrazin[1]

[1] UMR 5173 «Conservation des espèces, restauration et suivi des populations»
61 rue Buffon, 1er étage, 75005 Paris, France
[2] Service de Systématique Moléculaire (FR 1541), 43, rue Cuvier, 75231 Paris, France
[3] Service scientifique, Parc national des Pyrénées, 59, route de Pau, 65000 Tarbes, France
[4] Parc Régional du Vercors, 255, chemin des fusillés, 38250 Lans en Diois, France
[5] Division of Science & Conservation, Israel Nature & Parks Authority
3, Am Veolamo str., Jerusalem, 95463 Israel
[6] LPO PACA Antenne Verdon Canyon, 5, Boulevard Saint-Michel, 04120 Castellane, France
[7] LPO Grands Causses, le Bourg, 12720 Peyreleau, France
[8] Association Vautours en Baronnies, Mairie, 26510 Rémuzat, France
[9] Ornithological Station Cres, Institute of Ornithology
Ede Jardasa 35, HR-51000 Rijeka, Croatia
[*] Author for correspondence: E-mail: legouar@mnhn.fr

Abstract

Conservation actions and reintroduction programmes have been carried out to restore viable populations of Griffon Vulture in the South of France. Demographic and genetic studies were run to assess and understand the success of these programmes. In this paper, we focus on the genetics of these restored populations. Using micro-satellite markers, we investigated genetic diversity and structure of three native colonies that were spatially fragmented around the Mediterranean basin. We then assessed the genetic characteristics of four founder groups of reintroduction programs, and two settled reintroduced colonies in France.

We found that all studied populations of Griffon Vulture form only one genetically diverse unit, in which restricted gene flows between some colonies could lead to genetic differentiation. We thus propose that all Griffon Vulture colonies should be managed as one unit, optimising connections between them. Second, we found that random sampling of individuals among remnant populations of Griffon Vultures permits us to constitute highly diverse founding groups. Third, we showed that genetic diversity is preserved in the reintroduced colonies, probably because of high immigration rates of Pyrenean or Spanish individuals. We thus confirm that settled reintroduction has been a success from the genetic point of view. Finally, we highlighted a non-invasive method to genetically tag Griffon Vultures, which brings new prospects for monitoring and behavioural studies. Preliminary results on intra-population spatial structure, foraging behaviour and mating system are also presented.

Keywords: Griffon Vulture *Gyps fulvus*; Population genetics; Europe, France.

Introduction

Faced with the severe demographic decline and local extinctions of Griffon Vulture (*Gyps fulvus*) populations (Donázar, 1993), conservation actions and reintroduction programmes have been carried out to restore viable populations in the south of Europe. These actions were accompanied by monitoring of Griffon Vulture colonies, such as the census of breeding pairs and/or the ringing of individuals (Sarrazin *et al*. 1994, 1996; Terrasse *et al*. 2004). Thanks to this monitoring, actual demographic trends of Griffon Vulture populations in Europe are relatively well documented (see European census in Slotta-Bachmayr *et al*. 2004). In contrast, no monitoring of genetic diversity and structure of these populations has been carried out. However, genetic diversity is an indication of the species' evolutionary potential of adaptation to environmental changes (Frankel, 1974). Low genetic variability has been detected in some endangered vulture species (Bearded Vulture *Gypaetus barbatus*, Negro & Torres, 1999; Andean condor *Vultur gryphus*, Hendrickson *et al*. 2003; California condor *Gymnogyps californianus*, Ralls & Ballou, 2004). Major causes of this loss of genetic diversity could be population size reduction associated with genetic bottleneck, increase of inbreeding in small population and restricted gene flows in fragmented populations (e.g. Frankham *et al*. 2002). In the case of Griffon Vulture populations, we did not suspect that genetic factors could play a priori a negative role since the number of pairs in the biggest population in Europe (Spain) always remained up to 3,000 (Camiña, 2004), and no evidence of frequent expression of deleterious recessive allele had been noted. Nevertheless, we thought that genetic analyses of native and reintroduced Griffon Vulture populations would be useful to assess initial diversity, founder effects and gene flows. Moreover, knowledge of genetic diversity and structure of restored populations could be useful to assess success of restoration strategy. This genetic assessment of restoration programs could help to advise other conservation programs of this species or other similar species such as the Indian *Gyps* species which declined recently (Prakash *et al*. 2004).

We present a study combining genetic considerations in native populations and in pre and post-release phases of several reintroduction programs of Griffon Vultures in Europe. From the 80s to the present, several reintroduction programs of Griffon Vulture have taken place in France (see Terrasse in this issue for reintroduction details), inserted into a matrix of spatially fragmented native populations in the South of Europe. Using ten micro-satellite markers, we studied the genetic diversity and differentiation and migration rates of three native populations in order to assess if populations are genetically fragmented. We also assessed the genetic success of two reintroduced populations fixed for about 20 years, by evaluating its genetic diversity and its differentiation with native populations. Finally, we investigated the genetic diversity of four founder stocks of birds that were taken for the reintroduction projects. This study aimed at providing advice to future reintroduction projects.

Materials and Methods

Sampling

A total of 423 Griffon Vultures were sampled and analyzed. They belonged to three native colonies from Israel, Croatia, and Ossau valley in the French Pyrenees, two settled reintroduced colonies in France (Causses and Baronnies) and four founding groups of reintroduction programs (Navacelles, Baronnies, Verdon, Diois, Table 1).

Among the 96 analysed individuals released in the French Alps sites, 64 were known to come from Spanish rescue centres, and we used those individuals to test if Ossau could be considered as a sample of the Spanish population.

Molecular Techniques

DNA samples from Alpine sites, Navacelles,

Table 1. Number and type of samples by colony with indication on storage of sample until extraction.

Colonies		n	type	Storage until extraction
native populations	Israel			
	2000	35	adult feathers	Dry in paper
	Croatia			
	1997-2004	40	chick feathers	Dry in paper
	Ossau			
	1993	21	chick feathers	In 70% ethanol in tubes
	1996	29	chick feathers	In 70% ethanol in tubes
	2000	35	chick feathers	In 70% ethanol in tubes
settled reintroduced	Causses			
	1993	22	chick feathers	In 70% ethanol in tubes
	2000	41	chick feathers	In 70% ethanol in tubes
	2002	44	chick feathers	In 70% ethanol in tubes
	Baronnies			
	1999-2004	47	chick feathers	In 70% ethanol in tubes
founding groups	Navacelles	15	Blood	In tubes with lysis buffer*
	Baronnies	39	Blood	In tubes with lysis buffer*
	Verdon	25	Blood	In tubes with lysis buffer*
	Diois	30	Blood	In tubes with lysis buffer*

*lysis buffer: 0.1M Tris-HCl, pH 8.0; 0.1M EDTA, pH 8.0; 10mM NaCl; 0.5% SDS.

Grands Causses (1993, 2000) and Ossau were obtained using the Chelex 100 (Ellegren, 1994) and standard phenol/chloroform (Sambrook *et al*. 1989) protocols. For all other feather samples, we extracted DNA from a 2 mm piece of calamus with the CTAB method (Kretzmann *et al*. 2003). We retained ten micro-satellite loci for the quality of their amplification products and their polymorphism level on Griffon Vulture (Table 2). PCR (Polymerase Chain Reaction) reaction mixtures (10 µL final volume) contained approximately 2 ng of template DNA, 0.2 mM dNTP, 0.25 µM R-Primer, 0.25 µM L-Primer fluorescently labelled with one of 6-FAM, VIC, NED, PET (Applied Biosystems), 1 U Taq DNA polymerase and 1X buffer (Qbiogene). Cycling was performed in a MWG AG thermal cycler (Biotech) under the following conditions: 95°C for 5 minutes, 5 cycles of 1 minute at 95°C, 30 seconds at the suitable annealing temperature (from 53°C to 56°C) and 1 minute at 72°C, then 20 to 30 cycles of 30 seconds at 94°C, 30 seconds at the annealing temperature and 30 seconds at 72°C and a final step of 20 minutes at 72°C; annealing temperatures and cycle number depended on loci. PCR products were resolved on a 310 DNA sequencer and analyzed using GENESCAN software (Applied Biosystems).

Data Analysis

Genetic diversity and bottleneck detection:

We quantified the genetic variability within each group using GENETIX 4.02 software (Belkhir *et al*. 1996) and FSTAT 2.9.3.2 software (Goudet 1995). Departures from the Hardy-Weinberg equilibrium (HWE) were tested using GENEPOP 3.4 software (Raymond & Rousset 1995).

The probability of identity P(ID) is the probability that two individuals drawn at random from a population will have the same genotype at multiple loci (Taberlet & Luikart 1999). We computed the unbiased P(ID)

Table 2. The microsatellite loci and primer used GF loci have been characterized on Griffon Vulture by Mira *et al*. (2002), and GV loci on Bearded Vulture by Gautschi *et al*. (2000).

Locus	Fluorescent label	PCR Conditions	Primers (Forward, Reverse, 5'-3')
GF3F3	VIC green	55°C, 35 cycles	F GATCTTTCCCCTTCTGTG*
			R TTCGTGCAGTGATGCTGGTG
GF3H3	VIC green	56°C, 30 cycles	R GTAGAATAATTTGCTCCTGG
			F GTGAAGGCACCTCATAGACA*
GF8G1	PET red	55°C, 35 cycles	F TGAGCAGGTGAGTCCAGAAG*
			R GCTCTCCTGTCATCTTGCAT
GF9C1	6-FAM blue	53°C, 30 cycles	F GGTGGACATTACATACACTG*
			R CAAGGAATCTGGACTACTAA
GF11A4	6-FAM blue	53°C, 30 cycles	R GATCCCTTCCAACCGAAAAT
			F TGGTGACCAACGAAGTGTG*
GVBV11	PET red	56°C, 30 cycles	R TGTTTGCAAGCTGGAGACC
			F AAAAGCCTTGGGGTAAGCAC*
GVBV12	NED yellow	56°C, 30 cycles	R CCAGATAGGTTGGCAAAGATGC
			F GAAAGTGGTAACGGAGGAACAAG*
GVBV13	NED yellow	50°C, 30 cycles	F AAAACAGAGTTTTCACATTTTCATAAG*
			RTTCAGGAAACAGAAGCATGAAC
GVBV17	6-FAM blue	55°C, 35 cycles	F TGATGTGCAGATGCGTGAC*
			R GGACTCTGATGAAGCCAAGC
GVBV20	NED yellow	56°C, 30 cycles	F GAACAGCACTGAACGTGAGC*
			R GTTTCTCCTGACAGTGAAATAACTC

among the 423 analyzed individuals using formulae in Waits *et al*. (2001), implemented in the FAMOZ software (Gerber *et al*. 2003). The software calculated the unbiased P(ID) for each locus and classed them from the most informative [lowest P(ID)] to the least informative [highest P(ID)]. Adding locus in decreasing order of informative level, the software also computed unbiased P(ID) at multi-locus and allowed us to determine the number of loci necessary to differentiate all individual by a unique multi-locus combination.

We used the BOTTLENECK software (Cornuet & Luikart, 1996) to assess if the demographic decline of native populations, and the founder event of reintroduced ones, had impacted their genetic diversity.

Genetic differentiation and recent migration rates:

Wright's F-statistics: FST, FIS, and FIT across all groups as well as pairwise Fst were estimated according to Weir & Cockerham (1984) and the statistical significance of the FST was determined by 10 000 permutations with MSAnalyser 3.12 (Dieringer *et al*. 2002).

Population structure was inferred using GENECLASS2 software (Piry *et al*. 2004). The assignment test consists in calculating the likelihood of the multi-locus genotype of a given individual in a set of pre-determined samples. We used a criterion based on allele frequencies (Paetkau *et al*. 1995) to determine the most probable origin of each individual. The assignment of an individual to another sample than the sample where it comes from, suggests that gene flow has occurred between samples.

The genetic differentiation tests were first used to confirm if Ossau and Spain could be considered as one population, and then to determine structure in settled populations.

We estimated recent migration rates among native colonies and then among native and settled reintroduced colonies using the Bayesian inference method implemented in BayesAss 1.2 (Wilson & Rannala, 2003). We considered the most recent cohort of Ossau and Causses to estimate migration rates with Croatia and Israel. We excluded the wild born individuals of Baronnies because there is less than one generation time since the first reproduction in the wild in this colony.

Results

Polymorphism:

Polymorphism of the markers was high within populations. All loci were polymorphic in each group, except for Croatia for which

Table 3. Observed and expected probability of identity [P(ID)] for multilocus genotypes. Multilocus genotypes are constructed by adding loci in increasing unbiased P(ID).

Series of loci	Number of loci	Number of identical pairs of genotypes	Expected unbiased P(ID)
GVBV12	1	2314	0.02636
GVBV12+GF9C1	2	152	0.002
GVBV12+GF9C1+GF11A4	3	8	0.00009
GVBV12+GF9C1+GF11A4+GF3H3	4	2	0.00002
GVBV12+GF9C1+GF11A4+GF3H3+GVBV11	5	2	0.00002
GVBV12+GF9C1+GF11A4+GF3H3+GVBV11+GVBV20	6	1	0.00001
GVBV12+GF9C1+GF11A4+GF3H3+GVBV11+GVBV20+GVBV17	7	0	0
GVBV12+GF9C1+GF11A4+GF3H3+GVBV11+GVBV20+GVBV17+GF8G1	8	0	0
GVBV12+GF9C1+GF11A4+GF3H3+GVBV11+GVBV20+GVBV17+GF8G1+GF3F3	9	0	0
GVBV12+GF9C1+GF11A4+GF3H3+GVBV11+GVBV20+GVBV17+GF8G1+GF3F3+GVBV13	10	0	0

Table 4. Genetic variability for the 10 loci of the different groups of European Griffon Vultures: native and settled populations on the upper part and founding groups of reintroduction programs below.

Group	N	A	R	P	Ho	He	Departure from HW
Ossau total	85	6.0	3.8	100	0.5917 (0.2481)	0.5887 (0.227)	-
Israel	35	6.4	4.0	90	0.5299 (0.291)	0.5444 (0.292)	-
Croatia	40	5.4	3.8	90	0.5595 (0.263)	0.5797 (0.293)	GF11A4
Causses total	107	6.6	3.9	100	0.6036 (0.251)	0.5933 (0.224)	GF9C1
Baronnies wild born	47	5.7	3.7	100	0.6008 (0.24)	0.5732 (0.239)	-
Baronnies founder	39	5.5	3.7	100	0.5989 (0.259)	0.5708 (0.235)	-
Diois	30	5.4	3.7	100	0.5466 (0.264)	0.5527 (0.242)	GF8G1
Verdon Canyon	25	5.3	3.6	100	0.5465 (0.294)	0.5601 (0.243)	GVBV17
Navacelles	15	4.8	3.7	90	0.5336 (0.304)	0.5301 (0.299)	GF3H3

N: sample size, **A**: mean number of alleles per polymorph locus, **R**: allelic richness **P**: percentage of polymorph loci (95 % threshold), **Ho**: observed heterozygosity, **He**: expected heterozygosity at Hardy-Weinberg equilibrium, **Departure from HW**: locus showing a departure from Hardy-Weinberg equilibrium.

GVBV13 was monomorphic. The number of alleles per locus ranged from 2 for GF8G1 and GVBV17 to 21 for GVBV12. Probability of identity of loci ranged from 0.027 to 0.58 with highest values for loci with low number of alleles. Expected probabilities of identity for multi-locus genotypes were null from a combination of seven loci (Table 5). Thus, the diversity and the number of analyzed loci permit us to genetically characterize each individual analyzed by a unique multilocus genotype (Table 5).

Genetic diversity of settled populations:

Allelic richness ranged from 3.6 in Verdon Canyon to 4 in Israel (Table 4). Observed and expected heterozygosities ranged from 0, 5299 in Israel to 0, 6036 in Causses (Table 4) and were not different between settled groups. Each group was at HW equilibrium at each locus, except for GF11A4 locus for Croatia and for GF9C1 for Causses. All the groups can be considered as a pool of panmictic individuals. Genetic diversity, determined by the percentage of polymorph locus, the number of alleles per polymorph locus, and the heterozygosity rates, were similar within all the analyzed samples. No bottleneck was detected in native populations and settled reintroduced ones.

Genetic structure and recent migration rates between settled populations:

All structuring tests showed that individuals from Ossau and from the rehabilitation centre of Spain could be pooled in one population (F_{ST}=0 (ns) and 47% of individuals from Ossau were assigned to Spain whereas 50% of individuals from Spain were assigned to Ossau). So Ossau could be considered as a

Table 5. Percent of assigned individuals in each population (n=number of individuals assigned with % score more than 40%).

	Ossau	Israel	Croatia	Causses	Baronnies wild born
Ossau (n=74)	**46%**	4%	7%	19%	24%
Israel (n=28)	7%	**57%**	11%	14%	11%
Croatia (n=36)	8.5%	14%	**66.5%**	5.5%	5.5%
Causses (n=97)	17.5%	5%	4%	**54%**	19.5%
Baronnies wildborn (n=43)	28%	9%	5%	11.5%	**46.5%**

sample of the Spanish population.

Global F-statistics gave a low value of F_{IT} (total genetic variability, 0.02), with the principal part of variance being between populations ($F_{ST} = 0.019$). The F_{IT} value (within population variability) was only 0.001. F_{ST} calculated by pairs showed significant results between Croatia and all the groups. F_{ST} between Israel and Ossau 2000, Israel and Causses 2002 and Israel and Baronnies wild born were also significant. Little differentiation was also detected between Causses 2002 and Baronnies wild born. However, F_{ST} were very low (between 0 and 0.081).

Assignment tests showed that only 52.5% of individuals were well assigned to the population they came from (Table 5). Results of assignment test revealed that migration rates between settled colonies were frequent at least in the past.

However, with the Bayesian method implemented in BayesAss, we estimated restricted (but not null) recent gene flows between Croatia and Ossau, between Croatia and Causses, and between Israel and Causses. By contrast we detected higher migration rates from Ossau to Israel than from Israel to Ossau, higher immigration rate from Croatia to Israel than from Israel and Croatia and immigration rate from Ossau into the Causses colony was very high compared to the one from Causses into Ossau.

Genetic diversity of founding groups:

Observed and expected heterozygosities of founding groups were not different from native and settled reintroduced populations. Each group was at HW equilibrium at each locus, except for GF8G1 locus for the Diois, for locus GVBV17 for the Verdon and GF3H3 for Navacelles. In spite of the diverse origin of the founder no Walhund effect has been detected.

Discussion

Genetic diversity and structure in settled populations:

Our results suggest that, as expected, there are no genetic problems in Griffon Vulture populations. Indeed, we found no difference in genetic variability between settled populations. Allelic richness is the highest (with no statistical significance) in Israel, which can be explained by its position on the junction of migration routes, including vultures from Italy, Croatia, Serbia, Greece, Turkey, Jordan, Saudi-Arabia & Yemen populations. Furthermore, genetic diversity is quite high compared to that estimated for other species of vultures in Europe (*Gypaetus barbatus*, Gautschi *et al.* 2003, and *Neophron percnopterus*, Kretzmann *et al.* 2003). Demographic decline has not affected genetic variation of Griffon Vulture populations. Genetic diversity of reintroduced populations was as high as that one of native populations, which means that inbreeding had not occurred in these populations.

Assignment test revealed no structure among settled populations, which was in accordance with the high dispersal ability of Griffon Vultures. As a matter of fact, migration of juvenile Griffon Vultures to great distances is well known (Elosegi, 1989; Shirihai *et al*. 2000). Resightings of immature Griffon Vultures wing tagged in Croatia were mostly in the surroundings of Croatia (Italy, Austria, Greece), but occasionally Croatian vultures have been seen in Spain or Israel (Susic 2000). Beside observations, satellite tracking has also revealed that Griffon Vultures ringed in Israel could move to Finland, South Arabia, Turkey, Greece or Croatia (Hatzofe pers. comm.). The absence of structure revealed by assignment tests means that gene flows between studied colonies were historically frequent enough to consider these populations as one genetic unit. However, we detected surprising recent migration patterns. Indeed, asymmetrical migration rates seemed to be a frequent pattern between Griffon Vulture colonies. In particular, we found that recruitment of foreign birds to Croatia was low compared to departure of birds originating from the Croatian colony. One hypothesis is that effective dispersal between colonies was restricted by social behaviour. One specificity of the studied Croatian colony is that vultures there breed on cliffs above the sea, unlike most of the other European populations. Habitat selection of a long-lived colonial species such as the Griffon Vulture seems to be governed by a complex behaviour prone to both social and environmental interactions. “Cultural” transmission - which is based on non genetic differences between populations - could act as a brake to recruitment of exogenous birds and thus creates asymmetric gene flows between populations. The alternative scenario is that the Croatian colony has been recently cut off from the other populations by local habitat deterioration, leading to colony extinction. Study of fragmentation of a long-lived raptor, the Spanish Imperial Eagle, *Aquilla adalberti* showed that five to seven generations of isolation were enough to genetically differentiate populations (Martinez-Cruz *et al*. 2004). Indeed, disappearance of intermediate breeding areas in the French Alps occurred more than seven Griffon Vulture generations ago. Re-establishment of colonies in French Alps, Italy and Balkans should allow restoring of gene flow between Croatia and the other colonies.

Sampling strategy for future reintroduction program:

The random sampling strategy permits the creation of diverse founding groups, something that was theoretically expected when genetic diversity of the settled populations was high (Haig *et al*. 1990). The high genetic variability of founding groups limits any loss of genetic diversity expected with the founder event of reintroduction. Moreover, the little differentiation between settled populations would limit risk of out-breeding depression in the founding group. Consequently, if reintroduction is proposed in the Balkans and in North Africa, where local extinctions are always topical, we think that the random sampling of founders from among settled populations will be the optimum for genetic diversity.

DNA fingerprinting:

The ten microsatellite loci selected were highly polymorphic, allowing the characterization of each Griffon Vulture individual by a unique DNA fingerprint. This non-invasive method to genetically tag Griffon Vultures could be useful for monitoring the species. Indeed, colleagues have tested this. Feathers collected at feeding places make it possible to identify an individual previously sampled (Doxa pers. comm.). Unlike metal rings, genetic tags are permanent, and genetic monitoring allows minimizing the number of stressful captures in small and endangered populations (Rudnick *et al*. 2005). The DNA fingerprinting allows leading parentage analysis, which is useful to study mating systems (Ross, 2001). For example, we have analyzed parentage of newly reintroduced colonies in the Alps and it appears that Griffon Vultures are strictly monogamous. Studies using DNA fingerprinting would be very helpful to understand Griffon Vulture behavioural ecology such as habitat selection, which seems to be complex in this species.

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BREEDING SEASON RANGE USE OF A EURASIAN BLACK VULTURE *Aegypius monachus* POPULATION IN DADIA NATIONAL PARK AND THE ADJACENT AREAS, THRACE, NE GREECE

Dimitris Vasilakis, Konstantinos Poirazidis & Javier Elorriaga

WWF Creece, Dadia Project, GR-68400 Soufli, Greece
E-mail: ecodadia@otenet.gr

Abstract

In the framework of a sophisticated study for the Black Vulture *Aegypius monachus*, radio-tracking was employed to reveal range use patterns, identify areas of high risk and causes of mortality. Three immature and three adults were tracked during the breeding season in 2004. Six watersheds were distinguished covering the breeding colony and the foraging areas. Twelve permanent receiver stations were established on exposed hilltops. Vultures were tracked from three receiver stations, three days per week, each day in a different watershed alternating morning and afternoon sessions. Ninety-five percent minimum convex polygon (MCP) and fixed kernel (FK) range analyses were performed to estimate home range size and reveal range use patterns. Core areas were determined using incremental cluster analysis (ICP). The average home range was estimated with 95% MCP and FK as 91,300 ± 39,600 ha and 61,200 ± 22,400 ha, respectively. Inside the individual home ranges, 4-8 activity centres with an average total size of 30,700±5,600 km^2 were detected including on average 23% of the individual ranges. The overall area used by the six vultures, was estimated to be 178,200 ha when using the 95% MCP and 137,100 ha when applying FK. The vultures prospect a much wider area than Dadia NP during their foraging. The breeders had on average a smaller home range than the immature individuals and all of them showed colony-tenacity. Actions have to be taken to protect the foraging areas that were identified and to prevent (human) rural depopulation and the abandonment of traditional stock-raising practices.

Keywords: *Aegypius monachus*; Telemetry; Home range; Core areas; Range use; Greece.

Introduction

Animals commonly live in a spatially heterogeneous environment in terms of food availability, nest and roost sites, density of competitors, and other factors. The creation of space use maps from points representing distribution of animals or plants in space are critical in addressing a range of questions in ecology from the behavioural to the landscape level (White & Garrott, 1990; Kenward, 2001; Getz & Wilmers, 2004). Ecologists are generally interested in creating two types of such maps: home range maps that delineate the spatial extent or outside boundary of an animal's movement and utilization distributions (UDs) that represent the density of space used by animals and reveal range use patterns (White & Garrott, 1990; Getz & Wilmers, 2004). The depiction of the intensity with which animals use dissimilar parts of their home range presents an essential problem (Hodder *et al.* 1998). Most animals tend to occupy one or more certain areas (core areas) within their home range with higher frequency than other areas and it is possible that their behaviour will differ in the core and outer areas of the home range (Hodder *et al.* 1998). Several raptors and especially vultures are wide-ranging and their external home range limits may encircle habitat known to be avoided by them (Newton, 1979; Carrete & Donázar, 2005). Vultures are large scavengers that depend upon food resources that are sparse, patchily distributed and unpredictable in space and time and must travel extensively to forage successfully (Mundy, 1982; Maretsky & Snyder, 1992).

The Eurasian Black Vulture *Aegypius monachus* is listed as globally near-threatened (IUCN, 2003; http://www.redlist.org) and is considered as rare in Europe and endangered in Greece (Karandinos & Legakis, 1992). Greece is the only Balkan country holding a breeding population of this species which is located in the Dadia National Park (Dadia NP) (Hallmann, 1979; Handrinos, 1985; Poirazidis *et al*. 2004). According to the annual Black Vulture monitoring for the years 2000-2003, the number of breeding pairs ranged from 19-22 and the total population was estimated to be 90-100 (Skartsi & Poirazidis, 2002; Skartsi *et al*. 2003). The viability of the species' population in the area remains critical, as mortality factors like poison baits (Skartsi *et al*. 2003) continue to affect negatively the population and new potential threats have appeared after the recent establishment of wind farms (habitat degradation, possibility of collision) (Ruiz *et al*. 2005) in the vicinity of the nesting area and inside the foraging area of the population.

The conservation of this population is one of the central subjects of interest in the park management (Skartsi & Poirazidis, 2002). All conservation efforts so far have focused on the protection of the species' nesting habitat, supplementary feeding, the monitoring of the population and public awareness (Skartsi & Poirazidis, 2002; Poirazidis *et al*. 2004). Little attention has been paid so far to range use and movement patterns exhibited by this population, as well as to the kinds of dangers that it is facing during foraging. An understanding of these facts could be essential for the future management of this population at a larger scale.

Only a few studies have been carried out on the foraging habitat of the Black Vulture in Europe and none has attempted to identify areas of special importance for foraging and to produce relevant maps to promote the management and the conservation of the species in those areas (Corbacho *et al*. 2004; Carrete & Donázar, 2005). The present study aimed at promoting the conservation of this species in the Evros Prefecture and the adjacent areas by: a) determining individual home range size, b) revealing the internal structure of the home range, c) identifying areas of special importance, and d) estimating the overall area used by the population.

Study Area and Materials and Methods

The study area is located in the center of Evros Prefecture, it includes Dadia NP

(North-Eastern Greece, 26°00΄-26°19΄N, 40°59΄-41°15΄E), and is part of the southeastern tip of the Rhodopi mountain range, with altitudes between 10 and 1,200 m. The area is characterized by the interchange of small and large valleys, steep and shallow slopes, as well as an intricate hydrological network and is covered at low altitudes by extensive oak and pine forests, and at high altitudes by beech and oak forests. Many other habitats such as cultivations, fields, pastures and rocky slopes, are included. Vultures were trapped outside the breeding season (October and November 2003) (Skartsi & Poirazidis, 2002), using a walk-in cage with a sliding-door (Elorriaga *et al*. 2004) and a remote-controlled net trap (V. Matarranz, pers. comm.) baited with carrion. The vultures were aged (De La Puente & Elorriaga, 2004), sampled, measured and marked (Skartsi *et al*. 2003). Radio-transmitters (75gr TW3, Biotrack Ltd.) were attached on six vultures (Table 1) as backpacks (Bögel, 1994; V. Matarranz pers. comm.). In all cases the backpack/body weight ratio ranged from 1 to 2.7%, much less than the limit of 3% that is suggested for birds that migrate or depend greatly on flight (Kenward, 2001).

Six watersheds were distinguished in the study area, covering the breeding colony and the foraging areas. Twelve permanent receiver stations were established at exposed hilltops (altitude 200-1,100 m) overlooking the watersheds (Figure 1) from which extensive areas could be covered, thanks to the high transmission range of the transmitters (30–60 km, Biotrack Ltd.). Vultures were tracked from three receiver stations, three days per week, each day in a different watershed alternating morning and afternoon sessions in order to ensure that they would be tracked evenly throughout the study area, the seasons and the daylight hours. Radio-tracking teams used 4-element Yagi (Televilt Ltd.) antennas placed on 2.5 m poles and radio receivers R-1000 (Communication Specialist, INC). A compass rosette was used enabling us to eliminate the use of hand-held compasses, increased the accuracy and decreased the time needed for each triangulation (Vasilakis & Poirazidis, 2004). Bearings were obtained with standard radio tracking techniques (White & Garrott, 1990). An interval of at least 30 min was being allowed between successive location estimates on the same vulture to reduce dependency among estimates. It is commonly asseverated that for an unbiased estimate of the home range, independence of observations is required (Swihart & Slade, 1985; Harris *et al*. 1990; White & Garrott, 1990). However, several authors (Minta, 1992; Swihart & Slade, 1997; Otis & White, 1999) have argued that adequate sample size is more important than independence between locations. Eliminating auto-correlated locations from the data set, not only reduces the sample size, but it also reduces statistical power and the accuracy of home range analyses, and may also limit the biological significance of the analysis (De Solla *et al*. 1999). The inter-location interval of at least 30 min was considered as representative of a vulture's use of the study area, taking into account the flying abilities of the species. We used LOAS software (available at: http://www.ecostats.com/index.htm) to estimate vulture locations for sets of two or three simultaneous bearings. The point location estimates were computed using the magnetic declination (White & Garrott, 1990) of the study area (3.783°) and the evaluated method bias (0.53°) and standard deviation (9.68°) (Schindler *et al*. 2004). To calculate estimated locations the Andrews Estimator (White & Garrott, 1990) and the Best Biangulation were used. When three bearings were available, the locations were accompanied by an error ellipse varying in extent with the closeness of the intercepts (White & Garrott, 1990). We rejected all locations with ellipse size of more than 5,000 ha. In cases of two available bearings, triangulated locations composed of bearings at angles $<30^o$ or $>150^o$ were eliminated from all analyses (Zoellick *et al*. 2002).

The adequacy of radio-tracking sampling of individual vultures was assessed by relating

Table 1. Home range and core area estimates for the Eurasian Black Vultures *Aegypius monachus* during the breeding season (February-August) in Thrace, northeast Greece, using minimum-convex polygon (MCP), fixed Kernel method (FKM) and incremental cluster polygon analysis (ICP).

	IND	Sex	Age	Calendar year [a]	Sessions tracked	Sessions detected [b]	Total radio fixes	Sample radio fixes [c]	Home range estimates (ha)			Core area estimates (ICP)			
									100% MCP	95% MCP	95% FKM	Percent inclusion of fixes	Size in (ha)	% of home range	No. activity centers
	102	M	Im	4	52	44	187	133	190865	165408	96392	90	37603	19	8
	103	F	Im	4	52	49	321	254	174202	99747	74804	85	34857	20	8
	104*	F	Im	2	42	31	129	90	80905	53232	33957	95	22265	28	4
	105 [d]	F	Ad	>=6	52	43	203	156	137079	65641	49126	90	29422	21	5
	106 [e]	M	Ad	>=6	52	48	178	152	113472	81282	47797	90	32858	29	4
	107 [e]	F	Ad	>6	52	46	179	144	124171	82784	65143	85	27256	22	6
Mean (n=6)					50	44	200	155	136782	91349	61203	89	30710	23	6
SD					4	7	64	54	40373	39608	22391	4	5554	4	2
Mean Im (n=3) [f]							212	159	148657	106129	68384	90	31575	22	7
SD [f]							98	85	59264	56360	31709	5	8179	5	2
Mean Ad (n–3) [g]							187	151	124907	76569	54022	88	29845	24	5
SD [g]							14	6	11821	9494	9654	3	2825	4	1

IND, individual code; F, female; M, male; Im, immature; Ad, adult.

* The transmitter was fitted on the vulture on the 5th of May 2004, 3 months latter than the beginning of the breeding period.

[a] The age is referred to the breeding period and was determined based on the age of the individuals when they were trapped.

[b] Sessions with at least one successful triangulation.

[c] Number of fixes at which the home range was stabilized.

[d] Individual that started the breeding activities by constructing a nest.

[e] Successful breeders.

[f] Only the immature individuals were included.

[g] Only the adult individuals were included.

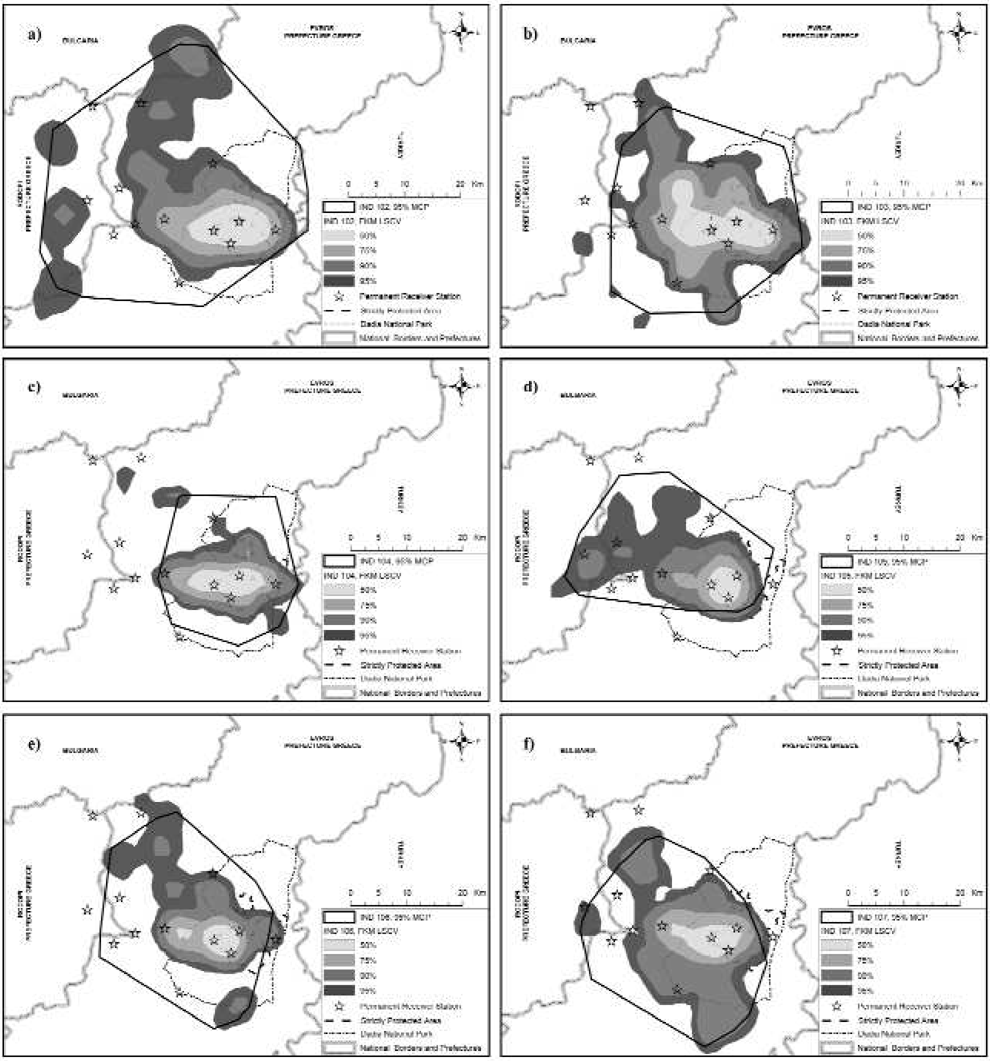

Figure 1. Home range estimates for the radio-tagged Eurasian Black Vultures *Aegypius monachus* during the breeding season. Solid line indicates the 95% Minimum Convex Polygon (MCP); graduated shaded polygons show the utilization distribution for fixed Kernel method (FKM). a) IND102, immature male; b) IND103, immature female; c) IND104, immature female; d) IND105, female breeder; e) IND106, adult male; f) IND107, adult female.

increase in home range size to successively larger sample of locations (Kenward, 2001). We determined the number of locations required to reach a range area asymptote by sequentially adding locations obtained for a particular individual, and then plotting range area versus the number of locations (Harris *et al*. 1990). We calculated home ranges and core areas of vultures for the breeding season (April–August 2004), using Arcview Version 3.2 with the extension 'Animal Movement' (Hooge & Eichenlaub, 1997) and BIOTAS (http://www.ecostats.com/index.htm). The ranges were estimated using the minimum-convex polygon (MCP) (Mohr, 1947), and fixed kernel method (FKM) (Worton, 1989). The 100% MCP estimates were used to facilitate comparisons between studies and regions. Ninety-five percent "peeled" MCP estimates were calculated using harmonic mean fixes as the range centre. We performed the FKM range analyses, because in addition to estimating range size, it reveals range use patterns, using a smoothing factor determined by least squares cross validation (LSCV) (Seaman & Powell, 1996). A grid cell of 2000x2000 m (400 ha) was used for FKM (Schindler *et al*. 2004) following the recommendation of Harris *et al*. (1990) that the grid cell size used is important as it reflects the accuracy of the fixes obtained by radio-tracking. Breeders were expected to spend increased amount of time on and/or around the nest (incubating, sanding and protecting the nestling) and in order to reduce the effect of tightly spaced locations on LSCV, in a radius of 1km around the nest and for each session only one estimated location was randomly selected and used. Core areas were determined by visually locating inflection points on utilization distribution (UD) plots (Harris *et al*. 1990). Incremental Cluster Polygon (ICP) is particularly suitable for this method because peeling of multinuclear ranges progressively excludes the smallest nuclei and tends to result in a 'steeped' UD plot (Kenward *et al*. 1998). The total area occupied by all the vultures was estimated by superimposing and merging the individual home ranges using the 100% MCP, 95% MCP and 95% FKM estimates. In this analysis the individual was used as a sampling unit. This technique is particularly suitable when inferences are going to be made about a population from a sample of individuals (Otis & White, 1999).

Results

The vultures (n=6) were monitored for 52 sessions from April till August 2004. The average number of sessions, when at least one reliable location per individual was obtained, was 44±7 (a±SD) (Table 1). The average home range size for all the individuals estimated with 95% MCP was 91,300±39,600 ha and with 95% FKM it was 61,200±22,400 ha. The immature individuals (i.e. No.s 102, 103 & 104) (Table 1, Figures 1a, b & c) had home range sizes which, on average, were bigger than that of the breeders (i.e. No.s 105, 106 & 107) (Table 1, Figures 1d, e & f), but the youngest immature (i.e. No. 104) appeared to have the smallest home range of all individuals. In the 95% MCP estimates, areas that were rarely used by the vultures were included, a fact showed by the FKM analysis. Additionally this analysis revealed the internal pattern of the home range and brought out areas often used outside the Dadia NP. A considerable overlap was detected also among individual ranges and all of them included a large part of the Dadia NP and extended to the northwest. An avoidance of the lowlands extending along the border between Greece and Turkey was observed (Figures 1 & 2).

The core areas were identified for every individual at 85-95% inclusion of locations (Table 1, Figure 2). Their average size was 30,710±5,554 ha and on average they covered 23% of the individual ranges. Core areas consisted of four to eight activity centres and considerable overlap was observed among individuals. The immature (4th calendar year, CY) individuals (i.e. No.s 102 & 103) included the same percentage of their ranges in the core areas and had the same number of activity centres (Table 1, Figures 2a & b). Additionally, they showed core areas larger than the other individuals. For the youngest (2th CY) individual (i.e. No. 104) the core areas were detected at 95% inclusion of locations, including the 28 % of its range which was 22,265 ha. Breeders showed similar-sized core areas with four to six activity centres. For the breeding females (i.e. No.s 105 & 107) the core areas represented a similar proportion of their ranges. It seems that the unsuccessful breeder (i.e. No. 105) undertook longer excursions during the breeding period in comparison with the successful breeders (Figure 2d). The overall area occupied was estimated as

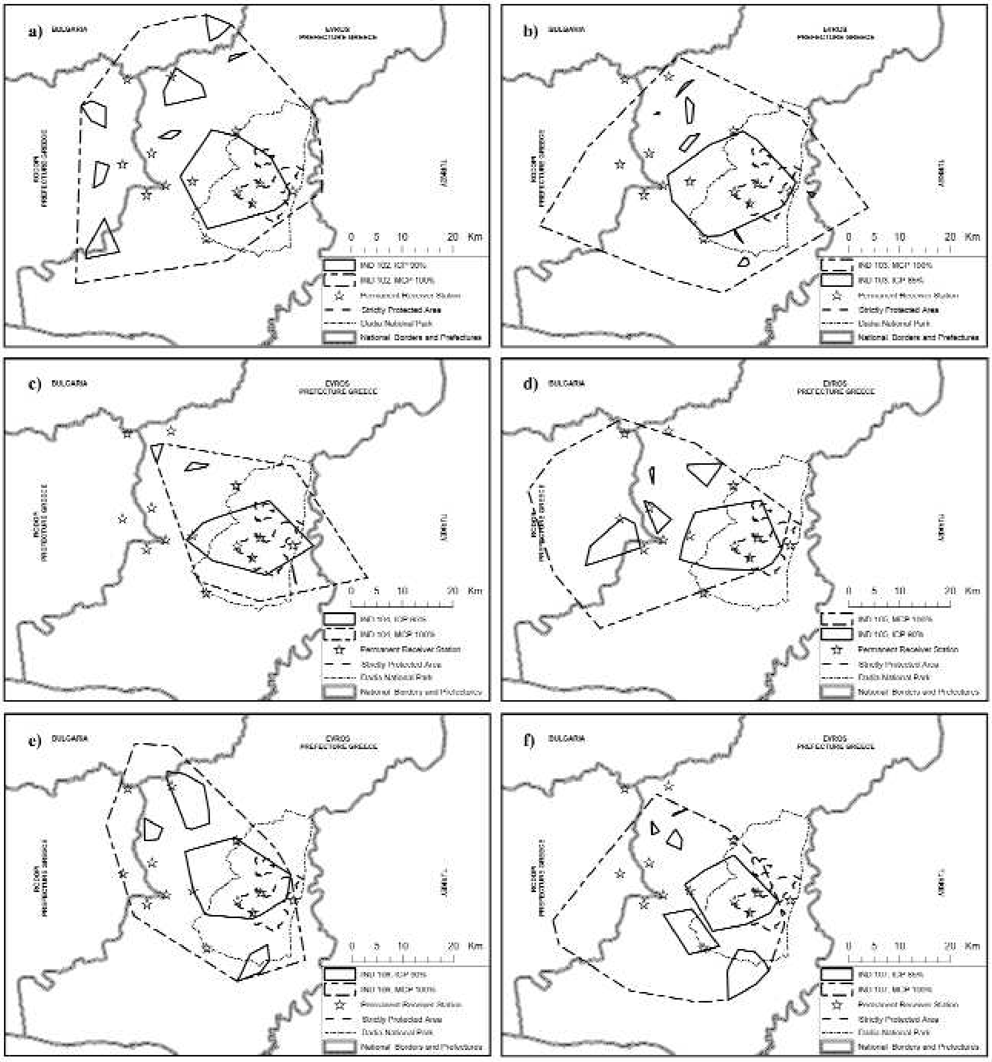

Figure 2. Core area estimates for the radio-tagged Eurasian Black Vultures *Aegypius monachus* during the breeding season. Hatched line indicates the 100% MCP; solid line shows the core areas with Incremental Cluster Polygons (ICP). a) IND102, immature male; b) IND103 immature female, c) IND104, immature female; d) IND105, breeding female; e) IND106, breeding male; f) IND107 breeding female.

243,326 ha with 100% MCP, 178,226 ha with 95 % MCP and 137,056 ha with 95% FKM. Areas with high use by the vultures were detected outside Dadia NP (43,000 ha) which constitutes only the one forth of the overall area used by all vultures during the breeding season (Figure 3).

Discussion

During foraging Black Vultures prospect a much wider area than Dadia NP. They search for food over large areas travelling far away from their familiar vulture restaurant in Dadia, visiting Bulgaria and the adjacent prefecture (i.e. Rhodope) encompassing within their

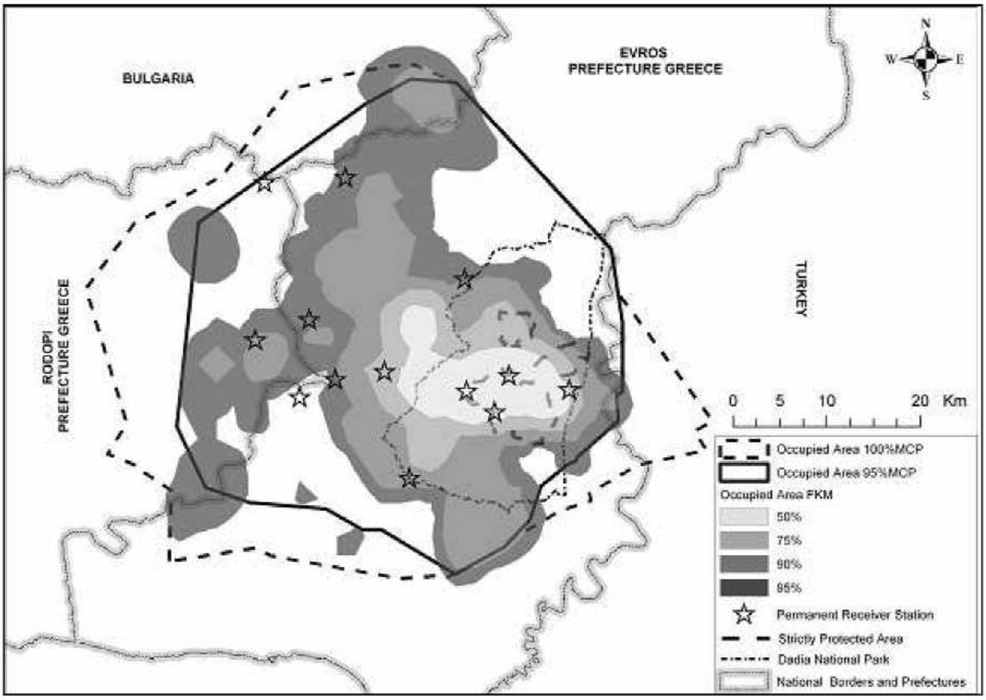

Figure 3. Overall area occupied by the radio-tagged Eurasian Black Vultures *Aegypius monachus* during the breeding season as it was determined by superimposing the individual home ranges (n=6). Hatched line indicates 100% MCPs estimates; solid line indicates 95% MCPs; graduated shaded polygons show the utilization distributions for fixed FKM.

home ranges thinly populated areas where traditional stock-raising practices are still common. During autumn, one adult breeding individual which had failed in incubation and an immature (1st CY) were observed 75 km far from Dadia, at a vulture restaurant in Studen Kladenetz, Bulgaria (Bulgarian Society for Bird Protection, unpublished data). In our opinion, these movements are very important for the survival and the expansion of the Black Vulture population and further research on these long distance movements could be extremely interesting.

Patterns of range use showed high variability depending on age and breeding status. Immature individuals showed a variable behaviour with frequent and long excursion movements and their foraging areas appeared to be bigger than those of the breeding individuals, which showed similar and smaller-sized home ranges. This can probably be explained by the fact that breeding individuals have to satisfy food provisioning for their offspring and themselves in the areas close to the nest site and this may force them to exclude long distance movements during breeding season. The avoidance of long distance movements by breeders is likely to be related to the excessive demands of incubation and the protection of their nestlings. The latter depend on the parental care (Donázar, 1993) for a prolonged period and probably there is a balance in time spent to take care of the nestling (thermoregulation, protection) and foraging (Morán-López *et al*. 2006) that affect the travel distances. The small home range size that was observed for the youngest immature could be probably explained by its inexperience that forced it to be attached to the natal area.

The home range (i.e. 76,569±9,494 ha) that we obtained for the breeders using the 95% MCP can be compared with the home range of 65,000 ha obtained with 95% MCP that is given by Corbacho *et al*. (2004) for a similar number of breeders at the Sierra de San Pedro colony (Extremadura, SW Spain) but for the annual home ranges. The home range estimate that we obtained for the three breeders with 95% FKM was 54,022±9,654 ha and is smaller than the 135,430±6,1191 ha that is given by Carrete & Donázar (2005) obtained with FKM for fourteen breeders of the Sierra Pelada colony (SW Spain) using on average 36±22 estimated locations per individual. It is possible that the availability of food, the disturbance in the foraging areas and the unsuitable habitats occurring between the colony and preferable habitats, are factors that affect travel distances and home range size and explain the differences between our estimates and those provided by Carrete & Donázar (2005). The difference between our estimate and that given by Corbacho *et al*. (2004) was less pronounced probably because of similarities in the two study areas.

The core area size showed a high variability for different individuals depending on their age but the main activity centre was common and included the area around the breeding colony. The Black Vulture as a semi-colonial bird seems to follow a central place foraging strategy (Carrete & Donázar, 2005) as the birds return every day to their nest or roost site in the colony. This was more evident for the youngest immature for which the area actively used for foraging was around the natal area. Activity centres were also detected outside the Dadia NP to the northwest of the colony. These areas could be important for the vultures as feeding and roosting grounds but also as areas where favourable conditions for thermal creation are available. It is interesting that one activity centre in the south of the park close to the breeding colony was used only by the breeders and not from the rest of the vultures. It seems that the breeders exploit the neighbouring areas more efficiently avoiding long distance movements.

In our study area the home ranges of the different individuals showed a considerable overlap. Furthermore, the individual ranges but also the overall occupied area showed an eccentrically spaced pattern around the colony with a prevailing northwest orientation. Taking into account the topography and the land use in areas in the east and southeast, this could be explained as an avoidance of the lowland intensively cultivated areas, where the absence of thermal lifts and the intensive human presence affect negatively the selection of these areas for foraging. It is possible that physiographic factors affect not only breeding habitat selection (Poirazidis *et al*. 2004; Morán-López *et al*. 2006) but also foraging habitat selection.

In conclusion, this study provides base line information on the range use and the movements of Black Vulture in Thrace. It is clear that Black Vultures include within their home ranges under-populated areas where pastoralism with traditional stock-raising practices is still common. Their movements and ranges expand inside Bulgaria where similar conditions prevail. The results of this study and the complementary spatial information have to be taken into account by decision makers and managers during rural development planning. Actions have to be taken, in order to enforce and broaden the trans-border collaboration on vulture conservation, prevent rural depopulation and abandonment of traditional stock-raising practices, stop the use of poisoned baits (Skartsi *et al*. 2003) and promote rural development plans that will safeguard the future conservation and expansion of this Black Vulture population.

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THE USE OF SOLAR-POWERED TRANSMITTING VIDEO CAMERAS FOR MONITORING THE BREEDING BIOLOGY OF BEARDED VULTURES *Gypaetus barbatus*

Antoni Margalida

Bearded Vulture Study and Protection Group
Ap. 43, E-25520 El Pont de Suert (Lleida), Spain
E-mail: margalida@gauss.entorno.es

Abstract

We designed a system of solar-powered video cameras that transmitted images via telemetry to a monitor. This system allowed us to study the breeding behaviour of the Bearded Vulture *Gypaetus barbatus* in the Pyrenees (NE Spain). From 2000-2006, 14 nests in 8 territories were equipped with video cameras. To avoid disturbing the birds, the equipment was installed 3-8 weeks before egg-laying. The acceptance rate was 78%. No decline in productivity was observed in the nests monitored with video cameras compared to control nests. The cameras enabled us to document egg-laying, hatching asynchrony, the nestlings' diet and the parents' breeding behaviour from distances of 2-3 km, although some technical problems temporarily interrupted the transmission of images. Video cameras can be used successfully to study this species at nesting cliffs, and probably other cliff-nesting raptors, without causing a decrease in productivity.

Keywords: Bearded Vulture; Cliff-nesting raptors; *Gypaetus barbatus;* Human disturbance; Pyrenees; Video cameras.

Introduction

Obtaining detailed data on the breeding biology for cliff-nesting raptors, including egg-laying, hatching asynchrony, diet, and causes of breeding failure, can be difficult due to the limitations of the location (inaccessible cliffs) and bird sensitivity to disturbance during the breeding period (see Richardson & Miller, 1997). In this sense, the use of video cameras in the study of the biology and behaviour of different species of raptors has increased over the last decade (Kristan *et al*. 1996; Delaney *et al*. 1998; Grønnesby & Nygård 2000; Dykstra *et al*. 2002; Booms & Fuller, 2003; Margalida *et al*. 2006). In the case of vultures, very little information is available on aspects of the birds' breeding biology in the wild (see revisions in Mundy *et al*. 1992), and in some species, the majority of the detailed information comes from captive individuals (e.g. Mendelsshon & Leshem, 1983). For threatened species, conservation priorities take precedence and thus human activities should be avoided in the area surrounding the nest (e.g. Steidl & Anthony, 2000). Such is the case for the Bearded Vulture *Gypaetus barbatus*, an endangered species, which inhabits European mountain ranges including the Pyrenees and the Alps (after their reintroduction in 1986), and the islands of Corsica and Crete. There are 122 breeding pairs in the European Union, 80% of which are in the Pyrenees (Heredia & Margalida, 2003). The Bearded Vulture is a territorial cliff-nesting accipitrid vulture whose diet basically consists of bones (Hiraldo *et al*. 1979). It is a long-lived species (Brown, 1997) characterized by late sexual maturity and a prolonged breeding cycle, beginning in September-October with the rebuilding of the nests (Margalida & Bertran, 2000b) and ending in June-July, when young fledge (Margalida & Bertran, 2000a; Margalida *et al*. 2003). Laying takes place in December-February and the incubation period is 54 days. The nestling period is about 4 months (Margalida *et al*. 2003). The Bearded Vulture's average productivity in the Pyrenees is less than 0.5 chicks/pair/year (Heredia & Margalida, 2001; Margalida *et al*. 2003), and appears to be very sensitive to human disturbance (Layna & Rico, 1991; Donázar *et al*. 1993).

We developed a radio-frequency-linked mini-camera system that transmits a video signal for documenting lesser known aspects of the Bearded Vulture's biology, which will help improve the application of conservation measures (e.g., rescuing the second nestling to increase productivity or to create a stock for captive breeding and studying the species' diet to improve the functioning of feeding stations). We tested this system during six consecutive breeding seasons, between 2000 and 2006, at 14 Bearded Vulture nests in a total of 8 territories. Herein, we describe the monitoring system (see also Margalida *et al*. 2006) and results after six years of study in order to analyze its advantages and disadvantages in their application for this and/or other vulture species.

Material and Methods

The study was in the Catalan Pyrenees mountains (NE Spain). This area contains 31 Bearded Vulture territories, of which 22 are breeding territories. The average maximum and minimum temperatures within the study area are 30ºC (July) and –5ºC (January), respectively. The average annual precipitation is over 800 mm, with 78 days of precipitation annually, which falls mainly as snow between December and February. The study area's terrain is rugged, which makes access to the nests difficult, and the average distance between nests and the nearest track is over 500 m. In the study area, the average elevation at which the Bearded Vultures nest is 1,387 ± 363.5 m (range 650-2,130 m, n = 48) and the average number of nests per territory is 4.7 ± 2.6 (range 2-11 nests, Margalida & Garcia, 2002).

The nests were located and monitored during September and October, when Bearded Vulture's nest-building begins (Margalida & Bertran, 2000b). Each year during the pre-laying periods (October-December), we installed two-three video camera systems in separate Bearded Vulture territories. The nests monitored with video cameras were situated at elevations between 900 and 1,650 m. The cameras were installed on the roofs of the cavities to provide a view of the inside of the nest and sufficiently high so as not to disturb the birds. The transmission system was installed on top of the cliff on eleven occasions and, on three occasions, at the bottom of the cliff. The average distance between the nest and the transmission system was 36.7 m ± SD = 21.1 (range 15-80 m, *n* = 12, Margalida *et al*. 2006).

The camera system (for more details see Margalida *et al*. 2005b, 2006) included transmitting equipment (a video camera and a transmitting antenna, powered by a solar panel or a wind-powered battery charger in one of the nests, and battery) and receiving equipment (a receiving antenna and a video recorder with a color monitor). The cost of the one complete system was approximately 4,200 €. The miniature video camera Panasonic measured 89 x 26 mm. It used a 12-volt power source (all components are 12 volt unless otherwise noted) and operated on a current of 100 mA. The camera was connected to a 2.4 GHz, water-resistant radio transmitter. The radio transmitter operated on a current of 180 mA. A small 50 x 10 mm microphone was connected to the transmitter. The camera and the transmitter were fixed to a wall using metal rock climbing materials (bolts). The transmitter was fixed using a 1-m aluminium support that allowed it to be pointed in the required direction (receiving equipment). The camera was connected to the transmitting antenna using a coaxial audio-visual cable, and the transmitter was connected to a power unit situated above or below the cliff using coaxial cable.

The power unit (battery) was attached to the support frame for the solar panels, which supplied the energy required by the transmission system. This power unit was a light sensor with a voltage of about 12 volts, and operated using a current of 5-60 mA. The light sensor was 65 x 45 x 30 mm and was connected to a solar regulator (P262-2), which operated on a current of approximately 0.1 mA (size 105 x 95 x 140 mm). It was connected to the Siemens solar panel with a nominal voltage of 15.5 volts, which were 1200 x 527 x 63 mm in size. This device charged the lead battery, and was 151 x 98 x 97.5 mm in size. The battery reserve capacity lasted for 3-4 days without sun and, if a deep discharge occurred, it took 3 hours of sunlight to recharge it completely. In one of the nests we used a wind-powered battery charger with an adapted volt regulator, which was 910 mm in diameter and 608 mm long, fixed to a 2m high mast. The image was received by an antenna programmed using the same frequency as the 2.4 GHz transmitter, which could be received at 1,000 m away. Line-of-sight was required between the transmitting and receiving antenna. The battery was powered using an audio-visual cable connected to one video recorder Sony mini DV format image receiver with a color, 148 x 62 x 135 mm LCD monitor, with its own battery or else connected directly to a 12-volt lead battery.

Results

From 2000-2006, 18 Bearded Vulture nests in a total of eight territories were monitored with video cameras. The equipment was installed 3-8 weeks before egg-laying. The average time it took to install the equipment in a nest for the first time, once we had reached the nest cliff, was 3.4 ± 1.2 h (range 2.2-6.4 h, *n* = 12, Margalida *et al*. 2006). This time depended on the cliff height and the climbing difficulty.

To monitor the 14 breeding attempts, a total of 18 camera systems were installed, which implies an acceptance rate of 77.8%. This was a result of some pairs changing nests. Of the 18 cameras installed, 12 were camouflaged with natural materials present in the nest (e.g., wool) and no camouflage was used with the remaining six. Although there are few data, it appears that the camouflaged systems were more readily accepted (83% vs. 50%).

We documented successful breeding at 9

(64.3%) of 14 camera systems that were installed and then accepted by the birds. In two of the remaining six cases, the pair did not lay eggs and in the third case the camera system was removed after the birds changed nest, for reasons probably unrelated to the presence of the cameras because this pair generally rebuilt several nests before egg-laying (pers. obs.). In this case, after the camera system was removed, the pair changed nest three more times and finally breed successfully. In the fourth case, during the incubation period the egg was rescued after it was seen that human disturbance was putting it in danger. This egg was incubated successfully in captivity; the chick hatched and then became part of the captive stock. In the remaining two cases, breeding failure takes place during the incubation period.

Productivity

The productivity of Bearded Vultures at occupied camera nests during the period 2000-2006 was 0.64 young/breeding attempt (n = 14) and at control (undisturbed) nests during the same period was 0.43 young/breeding attempt (n = 101). If we consider all the nests in which camera systems were installed (n = 18), the productivity was 0.5 young/breeding attempt. Thus, cameras did not negatively affect the reproduction of pairs. The cases of breeding failure documented (n = 5) took place during the incubation (three cases related to infertility of the eggs and nest-abandoning), hatching (one case) and chick-rearing (one case in which the chick died at age of 4 days).

System Performance and Problems

The camera system allowed us to document the egg-laying intervals (6 days on average, range 5-7, n = 7), time and incubation behaviour (53.4 days on average, range 52-55 and a median of prolonged incubation in the case of infertile eggs of 25 days, range 10-73, n = 10), hatching asynchrony (6.5 days on average, range 5-8, n = 6), sibling aggression (the age at which the second chick died varied from 4-9 days) and diet (for more details see Margalida *et al*. 2004, 2005). Image quality was good and only influenced by lighting conditions during direct sunlight (overexposure of the image made prey identification difficult). For example, the effectiveness of the system was demonstrated by the fact that that we were able to document the hatching interval (defined as the time elapsed in hours between the first observation of a hole in the egg until the time the chick was seen to be completely free of its eggshell) and all feeding bouts and aggression between siblings (Margalida *et al*. 2004). In addition, in three focal pairs in which a study of diet was carried out, 309 (87.8%) prey items could be identified out of 352 delivered to the nests.

During the second two weeks of December and the first two weeks of January (days with the lowest number of daylight hours), images could be received for 9 h (from 7.30 to 16.35) and 10 h (from 7.10 to 17.12), respectively. The number of hours during which the images were received increased as the number of daylight hours increased (e.g., 10.4 h during the first two weeks of February and 12.2 h during the first two weeks of March). Cloudy conditions influenced the number of daylight hours in which images could be received, varying between a few minutes and half an hour.

In two of the cameras, temperature fluctuations or precipitation caused condensation. Condensation occurred on the camera lenses during the months of December, January and March and between 11.00 and 16.00. Because this period coincided with the incubation period it was not possible to change the camera. In another camera, a mammal (possibly a Beech Marten *Martes foina*) chewed through the camera cable and disrupted the video signal for two months. The replacement of the camera was carried out during the second month of the nestling period, during which the chick remains alone for 15-20% of daytime (Margalida & Bertran, 2000a). In five cases, gentle movements were detected in the camera, which moved the lens focal point slightly. One of them was caused by the fact that the roof of the cavity was very close to the nest, which allowed the adults to go up to the camera and collect the wool camouflage for their nest. In the other four cases, the causes of the movement were unknown, although they might have been related to a problem with the camera ball-and-socket mount. In the first two cases, this occurred during the pre-laying period and could thus be corrected. In the third case, the movement was detected during incubation, but because it was very slight it did not prevent data from being gathered. In two more cases, movement of the focal point was detected during the nestling period. In one of them, we went to the nest while the adults were absent to correct the camera system, while in the other case it was not necessary to intervene. Finally, in two of the camera systems, the wind ripped off the solar panels, interrupting the transmission of the signal. These panels were replaced and it was then possible to continue the monitoring without any problems.

The system transmission was planned to be received within a 1-km radius. Nevertheless, the system was tested successfully at 3.5 km, and, although the quality of the images was lower, it was sufficient for the purposes of this study (e.g. to document laying intervals, hatching asynchrony), including prey identification. During the third year, several problems were detected in one of the transmitters, which were attributed to the material wearing out. This problem caused the transmission distance to be reduced to < 600 m. Wear on the batteries also caused problems and led to intermittent reception of images in two of the cameras during the third year of monitoring and in another camera during the fourth year. Low temperatures and constant recharging probably affected the life of the batteries.

Discussion

The results show that in the Bearded Vulture, a species very sensitive to human disturbance (Layna & Rico, 1991; Donázar *et al*. 1993), the final acceptance of the camera system apparently does not constitute an intrusive method of studying their breeding behaviour. The acceptance rate is higher than the rate of 65% obtained by Dykstra *et al*. (2002) for Bald Eagles *Haliaeetus leucocephalus*, a percentage considered by these authors as non-intrusive. Although some pairs changed nests, this behaviour has also been observed in undisturbed Bearded Vultures because this species can rebuild several nests before choosing the definitive site, so cameras probably have less effect than it might seem and are probably not responsible for the changes that were observed. In addition, in the study area the percentage of pairs that did not begin laying was on average 25% (*n* = 119 breeding attempts, Margalida *et al*. 2003), which means that the fact that they did not lay eggs was not necessarily directly related to the presence of the camera.

The effects of installing cameras in raptor nests during incubation or nestling periods vary from one species to the other. In the Bald Eagle, for example, results ranging from a high rate of nest abandonment (Cain, 1985) to successful nesting similar to those recorded in undisturbed nests (72% vs. 75% respectively, Dykstra *et al*. 2002). It appears that this species' reaction was related to the breeding period in which they were installed, the distance from the nest, or the birds' habituation to humans (Cain, 1985). However, in other species, such as the Peregrine Falcon *Falco peregrinus* (Enderson *et al*. 1972) and Osprey *Pandion haliaaetus* (Steidl *et al*. 1991; Kristan *et al*. 1996), no negative reactions were observed. The disadvantages of disturbance may be avoided or reduced by installing the system during the pre-laying period. In the case of the Bearded Vulture, nest-building behaviour takes place 2-4 months before egg-laying (Margalida & Bertran, 2000b), which facilitates locating nests well before the laying starts. Other advantages associated with installing the camera systems during the pre-laying period are: 1) it allows biologists to check whether the equipment disturbs the birds, and may allow the birds to become accustomed to the material and to accept its presence before breeding begins; 2) it allows biologists to check that the system works properly and leaves enough time for them to intervene if technical problems are detected or if the birds change nests. Moreover, the autonomy of our system allows the study to be carried out without having to visit the nest area after it has been installed (except in the case of technical problems). This reduces the potential negative effects the presence of a researcher would have on the breeding effort.

The camera systems permitted us to study aspects of Bearded Vulture's breeding behaviour (see Margalida *et al*. 2002, 2004, 2005) without causing a decrease in productivity. For example, for the study of the diet we identified 81.5% of the observed remains in the nest and 88% of prey delivered, a higher percentage than that obtained using telescopes (55.1% and 88.2% respectively, Margalida *et al*. 2005a). Similar results were obtained by Booms & Fuller (2003) in the Gyrfalcons *Falco rusticolus* (95%) with time-lapse studies in the same species. This percentage is important when considering that the Bearded Vulture brings in fragments of bone and half-consumed animal remains, which are very difficult to identify. Although this system involves a greater investment of time, because it means the researcher has to be present during the recording (the recording capacity of the tapes is only 90 minutes), it also has a series of advantages, such as: 1) a single receiving system can be used to monitor different nests, because it is easy to carry about; 2) it is no more expensive, since it avoids having to use and check countless tapes, as occurs in other types of studies; 3) it allows the interactions that occur around the nest (< 500 m) to be documented and this behaviour to be associated with what is going on inside the nest. Nevertheless, although the automatic recording system has also been used, a notable improvement in our system would be the replacement of the video-recording system by a computer hard disk (authors, unpubl. data) or a video-recording system that covered all daylight hours (> 14 h).

The disadvantages of the video system are mechanical failure, and the cost and time invested in monitoring. The problems related with mechanical failure can be resolved by increasing the capacity of the batteries, the size of the solar panels, the use of wind-powered battery chargers and repellents to avoid carnivores interfering with the equipment. In order to fix any technical problems without disturbing the birds, it is advisable to situate the transmission and power systems away from the nest and to ensure that they cannot be seen from the nest (installing them at the bottom of the cliff, for example). Regarding solar power, Booms & Fuller (2003) caution that this system may not be as reliable in non-arctic climates or in seasons when less sunlight is available. One solution to this problem, applied experimentally to the Bearded Vulture, is the installation of wind-powered battery chargers. These devices can replace the solar panels and can be especially effective in cliffs facing north, which receive little sunlight. In addition, wind-powered battery chargers permit the use of infrared cameras, allowing the batteries to be recharged at night. Concerning the time invested in monitoring, the use of other systems that permit 24 h recording time (e.g. Sony SVT-DL224 time lapse VCR, Booms & Fuller, 2003) would improve the efficiency of the system.

Although some camera systems can work perfectly for several years, it is wise to change the batteries annually and renew other parts such as the camera, the transmitter or the light sensor every three years. Thus, the cost of each system for monitoring other nests (the transmission alone) would be 1,950.00 €. The three-year renewal of the replacement components most likely to fail is about 825.00 €.

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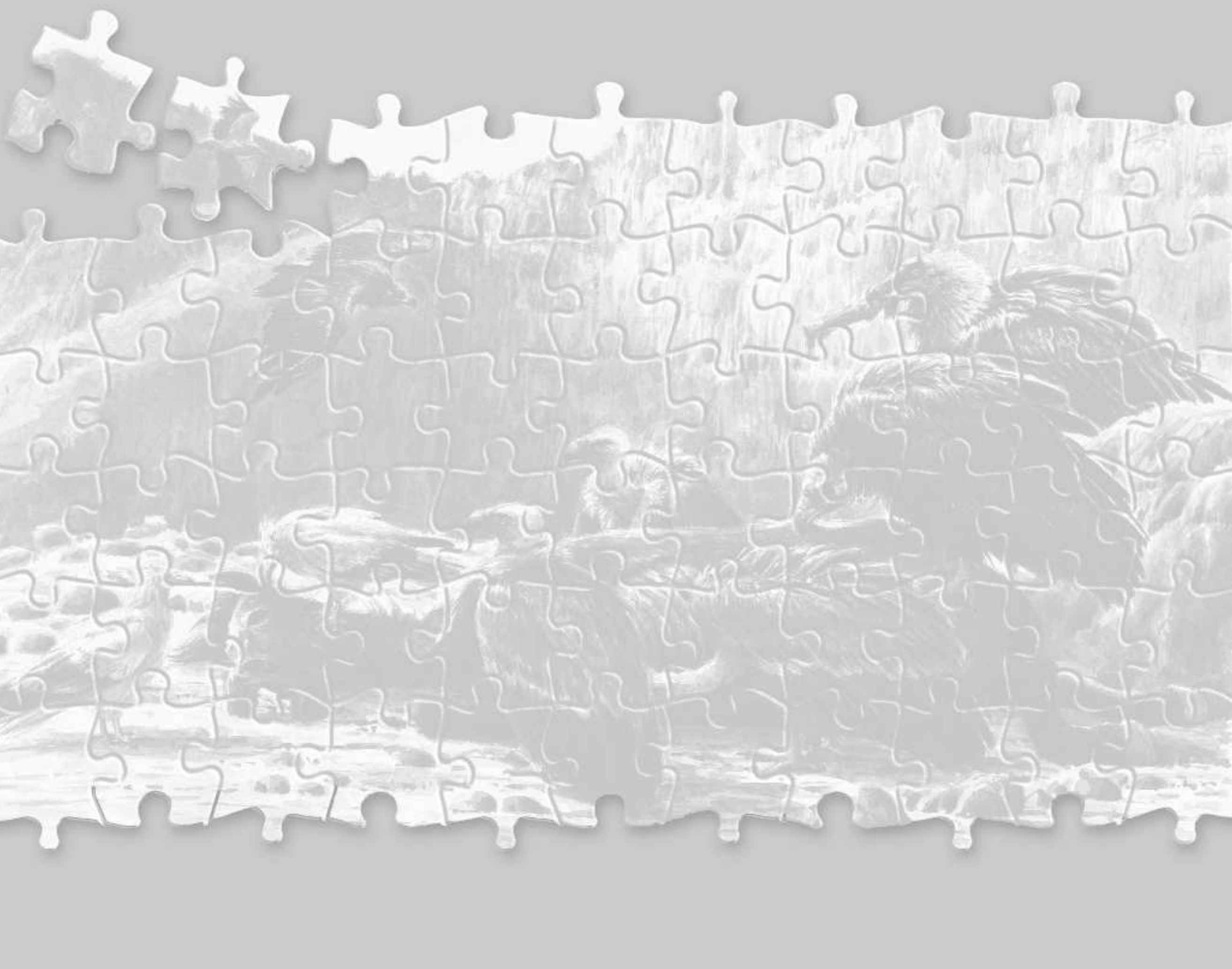
ANNEX I

ABSTRACTS OF PRESENTATIONS NOT SENT FOR PUBLICATION AS PAPERS



THE DESIGN AND IMPLEMENTATION OF TELEMETRY STUDIES: APPLICATIONS IN VULTURE SPECIES

Dr. Ofer Bahat

Department of Environmental Science and Chemistry
University of Indianapolis, Mar-Elias Campus, Ibillin, Israel
E-mail: bahat@shani.net

Abstract

Radio telemetry is a common tool in the study and conservation of birds of prey since the early 1970's. Vultures are distinguished carcass eating birds, and certain species are obligatory scavengers. Consequently, this unique diet, which is based on unpredictable food, results in foraging over extremely large areas. As a result, tracking vultures with the use of conventional radio-telemetry may be extremely difficult. One major obstacle in radio tagging vultures is their capture and recapture, which needs special expertise and experience to avoid damage or extensive stress to the birds. Nevertheless, radio-telemetry is a very important technique to collect various data as part of vulture studies or management and conservation programs. The radio-tracking project should be designed first to answer the biological questions or the aims of the management scheme. Secondly, it should meet the requirements of conservation and give information for management policies and activities when needed. A basic type of information obtained by radio-tracking vultures is their presence at a certain geographical area or site. A simple VHF receiver with scanner, even manually operated along constant intervals at a vulture nesting or roosting site, may be highly valuable for recording the presence of specific individual birds. The use of several tracking people may facilitate triangulation and enable to track birds over larger areas. Aircraft are highly efficient when there is a need to conduct telemetry searches over very large areas in a limited time. Conventional telemetry, once constant tracking of individual birds from dawn to dusk is achieved, may also be used to analyse time-budget activity. In addition to basic triangulation based on DOA (direction of arrival) antennas, automatic radio-telemetry stations were successfully used to track vultures, although they are expensive to purchase, they usually need specially designed transmitters and may cover a limited area of high probability receiving. Transmitters are usually attached to vultures in a backpack configuration, although patagial tags are also an option. In addition, leg mounts are useful when chicks are tagged at the nest to provide data on their post-fledging movements. Satellite telemetry was successfully used on vultures since the early 1990's. Although highly expensive and with limited accuracy and reliability, the Argos based PTT's were the source of invaluable data on vultures movements. GPS based satellite tags and modern data delivery systems (e.g. by GSM network) are important steps for significant improvement of data accuracy (including altitude) as well as reduction of costs of data acquisition.

Physiological data on vultures are also obtained by using radio-telemetry, mostly by implanting temperature and heart-rate measuring transmitters.

Important considerations related to vultures radio-tracking are the weight of a transmitter and harness and its possible effect on the wing loading of a species, and the tag shape, the extra drag it produces and its effect on the flight efficiency. This should be carefully examined, mostly when tagging relatively small sized vulture species.

FOOD EXPLOITATION BY GRIFFON VULTURES: THE EFFECT OF VULTURE RESTAURANTS IN SPAIN

Álvaro Camiña

ACRENA
Apartado de Correos 339, 28220 Majadahonda (Madrid), Spain
E-mail: acamia@vodafone.es

Abstract

The use of existing feeding sites (*muladares*), mainly by the Eurasian Griffon Vulture *Gyps fulvus* and other carrion eating birds, has been analysed in a vast area from Northern Spain, the Mountains of Sistema Ibérico that extends over more than 200 km in width. GLM models considering the amount of food available at the feeding site, numbers and ages of vultures present, distances to colonies, presence of other species and climatic variables were developed. The analysis also considered livestock density and forest cover on the estimated colony foraging range. The model explained a 65.23% of the variance. Number of immatures, the distance to the colony and the average colony size on a 15 km radius around the *muladar* were the only considered variables. The overall number of Griffons attending changed throughout the year (months) with a peak in June and lower use in January. Juveniles started appearing just after fledgling in July with higher numbers during August and September. Adult birds rarely increased above the 50% of the birds present. Immatures were scarce during laying and incubation periods (January-April) but greatly increased in May. Some feeding sites were not used from November to April with great amounts of carcasses accumulated without being consumed. On the other side a feeding site in Central-southern Spain revealed that large amount of juveniles gathered there for wintering with a lower use by breeding birds. This could suggest different strategies on food exploitation according to different food sources available (intensive vs. extensive farming). On the other side, large number of Griffons could be affecting by means of competition on other smaller carrion eating species. Management of feeding sites in Spain is rather complex than only supply a place with carcasses to fulfil both European and Spanish regulations. Attending on the ecology of vultures is essential for a proper management of the species.

ECOLOGICAL REQUIREMENTS OF REINTRODUCED SPECIES AND THE IMPLICATIONS FOR RELEASE POLICY: THE CASE OF THE BEARDED VULTURE

Alexandre H. Hirzel[1], Bertrand Posse[2], Pierre-Alain Oggier[2], Yvon Crettenand[3], Christian Glenz[4] & Raphaël Arlettaz[5]

[1] Laboratory of Conservation Biology, Department of Ecology and Evolution
University of Lausanne , CH-1015 Lausanne, Switzerland
[2] Bearded Vulture Network Western Switzerland, Nature Centre, CH-3970 Salgesch, Switzerland
[3] Game, Fishery and Wildlife Service, Canton of Valais, Rue de l'Industrie 14, CH-1950 Sion, Switzerland
[4] Laboratory of Ecosystem Management, Institute of Environmental Science and Technology
Swiss Federal Institute of Technology CH-1015 Lausanne, Switzerland
[5] Zoological Institute, Division of Conservation Biology, University of Bern
Baltzerstrasse 6, CH-3012 Bern, Switzerland, E-mail: raphael.arlettaz@nat.unibe.ch

Abstract

Species undergoing reintroduction offer a unique opportunity for clarifying their specific niche requirements because they are likely, if sufficiently mobile, to colonize the most suitable habitats first. Information drawn from the individuals released first might thus be essential for optimising species' policy as reintroductions proceed.

Bearded Vultures were extirpated from the European Alps about a century ago. An international reintroduction programme using birds reared in captivity was launched in 1986; up to 2003, 121 individuals had been released at four different locations. Subsequent dispersion throughout the range has been far from homogeneous, resulting in a clumped occurrence of the first breeding pairs within three main zones that do not necessarily coincide with release areas.

In order to discern ecological requirements we performed a geographical information system (GIS) analysis of Bearded Vulture sightings collected in Valais (Swiss Alps) from 1987 to 2001. This area harbours no release site, is situated in the core of the Alpine range and has been visited by birds from all four release points.

During the prospecting phase (1987–94, mostly immature birds), the most important variable explaining bearded vulture distribution was ibex biomass. During the settling phase (1995–2001), the presence of birds (mostly maturing subadults) correlated essentially with limestone substrates, while food abundance became secondary.

The selection of craggy limestone zones by maturing Bearded Vultures might reflect nesting sites that are well protected against adverse weather, as egg laying takes place in the winter. Limestone landscapes, in contrast to silicate substrates, also provide essential finely structured screes that are used for bone breaking and temporary food storage, particularly during chick rearing. Finally, limestone substrates provide the best thermal conditions for soaring.

Extrapolated to the whole Alpine range, these findings might explain both the current distribution of the subadult/adult population and the absence of breeding records for Bearded Vultures around release sites in landscapes dominated by silicate substrates. As reintroduced Bearded Vultures tend to be philopatric, we suggest that population restoration would be more efficient if releases were concentrated within large limestone massifs. This case study of the Bearded Vulture illustrates the need for continual adaptive management in captive release programmes.

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MODELLING VULTURE HABITATS IN THE CAUCASUS

A. Gavashelishvili[1] & Michael J. McGrady[2]

[1] Georgian Centre for the Conservation of Wildlife (GCCW)
P.O. Box 56, Tbilisi 0160, Georgia
E-mail: kajiri2000@yahoo.com
[2] Natural Research, Ltd., Am Rosenhügel 59, A-3500 Krems, Austria
E-mail: mikejmcgrady@aol.com

Abstract

Breeding Habitats

Nest-site selection was examined by comparing habitat variables at nest-sites occupied by vultures with those at potential but unoccupied sites. Identification of potential nest-sites was based on the cover they offered from climatic adversity, direct human approach, and predation. Predictive models of nesting site selection were estimated using logistic regression procedure.

The probability of Bearded Vulture occupancy of a cliff ledge that was safe from climatic adversity, direct human approach, and predation was negatively correlated with road network density and positively correlated with elevation, area steepness, percentage of open areas, number of wild goats and annual biomass of dead livestock.

The probability of a safe cliff ledge being occupied by Griffon Vulture was negatively correlated with annual rainfall, and positively correlated with the percentage of open areas and annual biomass of dead livestock.

As for Cinereous Vulture, the best model suggested that in Georgia a 20 x 20-m plot was more likely to contain a Cinereous Vulture nest if the slope was > 30^{o} and faced north rather than south, was situated in rugged terrain away from unprotected and populated areas and was relatively dry.

Foraging Habitats

The study of habitat variables influencing the response of Griffon, Cinereous and Bearded Vultures to carcass appearance showed that Cinereous Vultures were the first to arrive at a carcass, followed by Griffon Vultures, then Bearded Vultures.

The probability that all three species would land and eat a carcass was positively correlated with the extent of visibility around the carcass and negatively correlated with road network density.

Besides, the probability that Bearded Vultures would feed at a carcass was positively correlated with area steepness around the carcass. The smaller the carcass and farther from the nearest populated area and Bearded Vulture nest, the sooner Bearded Vultures fed at it.

The probability that Griffon Vultures would feed at a carcass was positively correlated with proximity to populated areas, area steepness and the percentage of open areas. The larger the carcass and farther from the nearest point of roads, the sooner Griffon Vultures fed at it.

The probability that Cinereous vultures would feed at a carcass was negatively correlated with annual rainfall, and positively correlated with the percentage of open areas. The farther from the nearest populated area the carcass was, the sooner Cinereous Vultures fed at it.

DYNAMIC OF RESTORED POPULATIONS OF GRIFFON VULTURES IN SOUTHERN FRANCE

F. Sarrazin[1], M. Bosè[1], A. Gault[1], P. Le Gouar[1], A. Robert[1], C.P. Artur[2], J.P. Choisy[3], S. Henriquet[4], D. Peyrusqué[2], J.L. Pinna[5], P. Lecuyer[6] & C. Tessier[7]

[1] Lab. Conservation des Espèces , Restauration et Suivi des Populations (CERSP)
UMR 5173 MNHN-CNRS-UPMC, 61 rue Buffon, 75005 Paris, France , E-mail: sarrazin@mnhn.fr
[2] Parc National des Pyrénées
[3] Parc Naturel Regional du Vercors
[4] LPO PACA
[5] Parc National des Cévennes
[6] LPO Grands Causses
[7] Vautours en Baronnies

Abstract

The restoration of Griffon Vulture populations has been particularly efficient in Southern France where a natural population in the Pyrenees (Ossau Valley) and five reintroduction programmes (Causses, Navacelles, Baronnies, Verdon and Vercors) have been monitored over the last decades. This monitoring based on marked individuals and breeding pairs allowed us to understand the demographic processes underlying this restoration. Survival, reproduction and dispersal were estimated in an integrated and comparative approach to assess the natural dynamics of Griffon Vulture populations, the short and long term effects of reintroductions on these dynamics and their likely regulation through density dependence. Feather or blood samples were collected on more than 850 wild born or released individuals to allow molecular sexing. These samples were combined with biometric measurement. Survival analyses involved combination of capture-mark-resightings and recoveries in multistate models to account for ring losses. Breeding parameters were estimated in each population through regular monitoring during the breeding season and GLM analyses. Dispersal was studied using both genetic and demographic approaches. Population dynamics were investigated using structured population models. Additional studies of population genetics (see Le Gouar *et al*. this volume) and of competition for food were run to complement our understanding of these demographic processes.

Overall, annual survival and particularly adult survival remained remarkably high over more than two decades showing the relevance of conservation measures on this parameter which contribution to long lived population's growth rate is of first importance. No bias in sex ratio as well as no effect of sex on biometry, survival and dispersal was detected in both released and wild born birds. Despite initial survival cost repeatable in the released populations, high following survival, good breeding success and efficient immigration induced a strong dynamics in both natural and reintroduced populations. Preliminary comparative analyses of Causses and Ossau breeding success showed an effect of density in the Ossau colony together with a reduction of body size index in nestlings. Behavioural analyses of foraging strategies and feeding events in the Causses showed that adults were highly dominants and that feeding places had no strong negative impact when they allow spatio-temporal stochasticity in food availability. The restoration of a food management involving numerous farmers appears thus to be a good compromise to reduce intraspecific competition and insure a long-term integration and viability of Griffon Vultures in human landscapes.

ACTIONS FOR THE REINTRODUCTION OF THE BEARDED VULTURE *Gypaetus barbatus* IN ANDALUSIA

Sergio Couto

Fundación Gypaetus
C/Cataluña 8, 1A, 23009 Jaén, Spain
E-mail: sergiocouto@gypaetus.org

Abstract

The European LIFE project "04NAT/ES/000056" has the goal of establishing a viable population of the endangered Bearded Vulture *Gypaetus barbatus* in Southern Spain, where this species inhabited until the ‘80s. All actions will be the result of the multiple joint efforts that Andalusian Regional Government and Gypaetus Foundation have been developing since 1996. Current partners of the LIFE are Andalusian Hunting Federation, the Andalusian Environmental Ministry and the Small Farmers and Stockbreeders Association. The project has five main working lines (following IUCN guidelines) that should allow this reintroduction: (1) assessment of habitat feasibility; (2) captive breeding; (3) removal or control of past and present dangers for the species survival (i.e. illegal poisoning, power lines, illegal hunting); (4) environmental education; (5) release of specimens and restoration of the population. The Breeding Centre is located in the Sierra de Cazorla, Segura y Las Villas Natural Park. The specimens were provided by the Foundation for the Conservation of the Bearded Vulture, Conselleria d’Agricultura, Ramaderia i Pesca of Generalitat de Catalunya and Diputación General de Aragón. This centre belongs to the Endangered Species European Program for the Bearded Vulture (EEP). Twenty-seven specimens form the breeding stock (13 males, 11 females and 3 juveniles). Five of them are founder individuals and the other belong to F1, F2, F3. The specimens are of Pyrenean, Asiatic, continental Greece and Crete origin. Molecular (mt DNA and microsatellite) analyses support a reintroduction based on descendants of this breeding stock. Specimens will be released and radio-tracked using GPS emissors. Poisoning and power lines are major threats for the survival of these specimens. Power lines have been studied to propose management measures. Moreover, the war against poisoning is now very intense in Andalusia. The Andalusian Environmental Ministry together with the Gypaetus Foundation is now working on 61 actions against poisoning. The most relevant are: analysis of the distribution of poisoning activities; control of the distribution of chemical substances used as poison; search for poison bats by trained dogs; fine and application of disciplinary measures for illegal activities. Simultaneously, we are carrying out environmental education campaigns together with the release of diverse explicative publications about the programs, mainly directed toward children, hunters, stockbreeders and distributors of chemical substances. These activities for the conservation of the Bearded Vulture will benefit other autochthonous species and contribute to the restoration of ecological processes in Mediterranean mountains.

USING MICROSATELLITE MARKERS TO INFER THE GENETIC STRUCTURE OF *Aegypius monachus* (AVES: ACCIPITRIDAE)

N. Poulakakis[1], G. Mantziou[1], A. Antoniou[2], A. Parmakelis[1], Th. Skartsi[3], D. Vasilakis[3], J. Elorriaga[3], J. de la Puente[4], A. Gavashelishvili[5], M. Ghasabyan[6], T. Katzner[7], M. McGrady[8], N. Batbayar[9], M. Fuller[10] & T. Natsagdorj[11]

[1] University of Crete – Natural History Museum of Crete, Laboratory of Molecular Biology
P.O. Box 2208, GR-71409 Heraklion, Crete, Greece, E-mail: poulakakis@nhmc.uoc.gr
[2] Genetics and Molecular Biotechnology Department, Hellenic Centre for Marine Research – Crete
Thalassocosmos (Former U.S. Base at Gournes), P.O. Box 2214, GR-71003 Heraklion, Crete, Greece
[3] WWF Greece, Dadia Project, Dadia, GR-68400 Soufli, Greece, E-mail: ecodadia@otenet.gr
[4] Área de Estudio y Seguimiento de Aves, SEO/BirdLife
Melquiades Biencinto 34, E-28053 Madrid, Spain, E-mail: carricero@worldonline.es
[5] Georgian Centre for the Conservation of Wildlife (GCCW)
P.O. Box 56, Tbilisi 0160, Georgia, E-mail: kajiri2000@yahoo.com
[6] Armenian Society for the Protection of Birds, 16 Block, h.30/126, 375115Yerevan, Armenia
[7] E-mail: tkatzner@imperial.ac.uk
[8] Natural Research, Ltd., Am Rosenhügel 59, A-3500 Krems, Austria
[9] Wildlife Science and Conservation Centre of Mongolia
Bldg. Institute of Biology, Room 309, Ulaanbaatar 210351, Mongolia
[10] USGS, Forest and Rangeland Ecosystem Science Center, Snake River Field Station, and Boise State University – Raptor Research Center, 970 Lusk St., Boise, Idaho 83706 U.S.A.
[11] Ornithological Laboratory, Institute of Biology
Mongolian Academy of Sciences, Ulaanbaatar 210351, Mongolia

Abstract

Considering the severe decline that populations of the Black Vulture *Aegypius monachus* have experienced during the last century, we employ nuclear molecular markers (microsatellites) in an attempt to investigate the population status and the current genetic diversity for most of the species populations. Samples were collected from 6 different countries representing most of the species distribution and genotyped for 8 microsatellite loci originally developed for 2 other vulture species. Results indicate that most of the analysed populations departed from HW equilibrium as a result of significant heterozygote deficiency. Genetic differentiation between pairs of populations was statistically significant for most of the analysed populations. We obtained medium to high Fst values that correspond to the geographic distances of the populations under study. Patterns of population structure were investigated using a Bayesian clustering approach that revealed the uppermost hierarchical level of population structure while subsequent analyses of the defined subsets allowed finding the within-group genetic structure. Further investigation of these populations is required in order to draw any conclusions about future conservation practices.

THE GENETIC STRUCTURE OF THE BEARDED VULTURE *Gypaetus barbatus* POPULATION IN CRETE

Nikos Poulakakis, Georgia Mantziou & Aris Parmakelis

University of Crete – Natural History Museum of Crete, Laboratory of Molecular Biology
P.O. Box 2208, GR-71409 Heraklion, Crete, Greece
E-mail: poulakakis@nhmc.uoc.gr

Abstract

European populations of Bearded Vulture *Gypaetus barbatus* have experienced a severe decline during the last century. This study is an attempt to investigate the population status and the current genetic diversity of the population in Crete. Fifteen samples from Crete are molecularly sexed and genotyped for eight microsatellite loci. The results indicate that the population of Crete is divided into two subpopulations. Bayesian approach identifies the two populations clearly and exact tests showed significant differentiation between them. Phylogenetic trees were reconstructed with the use of Neighbour Joining method with genetic distances DAS and Dc. We obtained medium to high Fst values between the populations that indicate a low gene flow between them.

DISCRIMINATION OF THE SEX IN THE CINEREOUS VULTURE *Aegypius monachus* USING MORPHOMETRIC TECHNIQUES

Javier Elorriaga, Kostas Poirazidis, Dimitris Vasilakis & Theodora Skartsi

WWF Greece, Dadia Project
Dadia, GR-68400 Soufli, Greece
E-mail: ecodadia@otenet.gr

Abstract

Sex identification of birds is essential as base of ecological research and conservation. The development of molecular techniques based on DNA analyses allows researchers to identify the sex of birds in a highly accurate way. Nevertheless, molecular sexing requires a considerable financial investment on laboratories and frequently a long time of waiting between sampling and sex determination. Many ornithological studies require a field technique for immediate sex identification. In many bird species, techniques for in-hand sex determination based on the species biometry have been proposed. Although the existence of slight "reversed size dimorphism" among European vultures is nowadays accepted, different authors consider that sexual-size differences in the Cinereous Vulture are not significant due to the high degree of overlap in the analysed measurements. However, the available data on this subject are scarce.

In this work we studied the biometry of Cinereous Vultures from Dadia National Park (Greece) focusing on sex-related differences in order to develop a reliable technique for sex determination in the field. The study was based on a sample of 32 free-living Cinereous Vultures from which 16 biometric measurements (variables) were collected: Primary 9 (P9), Primary 8 (P8) and Primary 7 (P7), Wing Chord (WC), Central Rectrices (CR), Central Toe (CT), Central Nail (CN), Tarsus Height (TH), Tarsus Width (TW), Tarsus Length (TL), Bill Width (BW), Bill Height (BH), Bill Length (BL), Exposed Cullmen (EC), Head Length (HL) and Head Width (HW). Vultures were aged in calendar years according to their moulting pattern and grouped as adults (?5 CY) or immatures (<5 CY). Sex was identified with molecular techniques. Those variables that showed low repeatability were excluded (P9, P8, P7, WC, CR, CN, TL). The remaining measurements revealed a more accurate "reversed size dimorphism" in immatures than in adults. Using correlation tests the highly correlated variables were excluded (TW, BL, EC). MANOVA showed size differences between immatures and adults. ANOVA indicated that those differences were significant in BW, BH, HW and HL and not significant in CT and TH. Thus, the first four variables were age-dependent and the last two age-independent. Stepwise Discriminant Analysis was used to discriminate the sexes. The best predictor variables were TH among the age-independent and, among the age–dependent, HL and BH for adults and BW for immatures.

IMPACTS OF WIND FARMS ON BIRDS IN EVROS AND RHODOPI, GREECE: PRELIMINARY RESULTS

Carlos Ruiz González, Kostas Poirazidis & Stefan Schindler

WWF Greece, Dadia Project
Dadia, GR-68400 Soufli, Greece
E-mail: ecodadia@otenet.gr

Abstract

According to European policy, wind farms are a promising alternative energy source. Wind farms can be considered as "green" energy but in some cases, especially in areas where protected species of fauna occurs, some problems can rise. In this study the impact of two wind farms on a raptor community in northeastern Greece is presented.

The study area is located in Evros and Rhodopi prefectures, Greece, close to the last breeding colony of Eurasian Black Vulture *Aegypius monachus* in the Balkans (National Park of Dadia-Soufli-Lefkimi).

The main objective was to determine the impacts of these wind farms on the raptor population. Is focused on: (1) death of birds through collision with the turbines; (2) evaluation of risky behaviour and risk factors; (3) changes in the use of the habitat and evaluation of habitat loss due to the wind farms, and (4) proposing measures to reduce the impacts once the risk factors are detected.

Two viewpoints were selected in each wind farm, overlooking the major part of the turbines. We visited the viewpoints during 52 days of fieldwork, 4 hours each time. In addition, a fix group of turbines was walked in every visit to search for dead birds.

The data collected include species, number, sex, age, type and height of flight and activity of all observed raptors. Additionally if the birds had been observed close to the turbines or crossing the "row", some other variables has been collected, like the closest turbine, the distance to the turbine, etc.

The results of this study cover the breeding season, summer and autumn 2004.

No raptors were found dead. Because of this, it is impossible to relate risk events directly with mortality and a low mortality can be expected. Most of the birds detected in the risk area were local birds. High proportion of detected birds in the risk area crossed the turbines, thus the effect of the windmills on the behaviour of the birds is supposed to be rather low. Many movements of the raptors occurred close to the outermost turbines, but many vultures were observed to cross the lines in the gaps of the lines.

EVALUATION OF THE USE OF MICROCAMERAS IN NEST MONITORING OF THE BEARDED VULTURE IN CRETE

Costas Grivas[1], Costas Christodoulou[2], Stavros Xirouchakis[3] & Michalis Probonas[4]

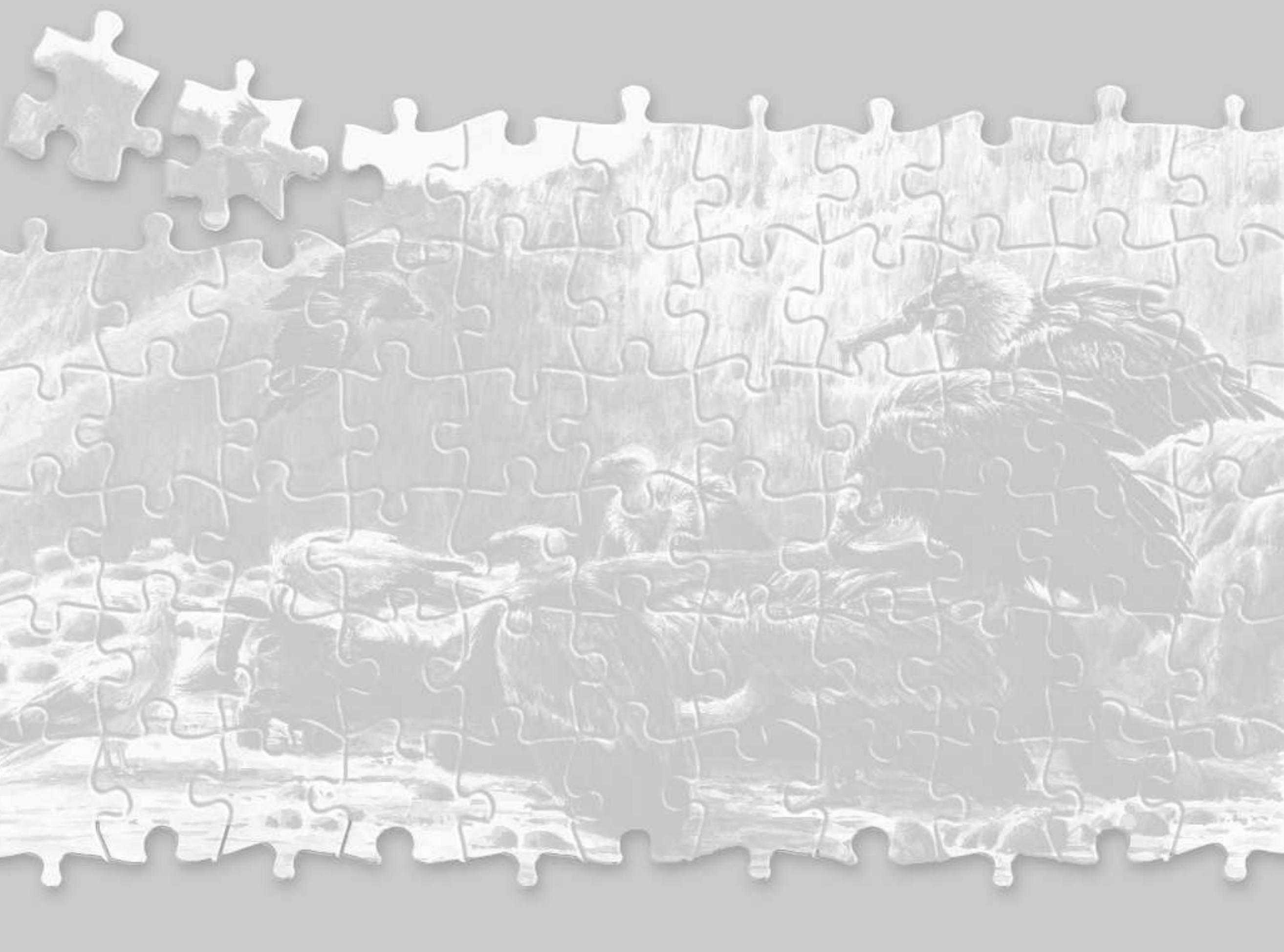
[1] University of Crete – Natural History Museum of Crete
P.O. Box 2208, GR-71409 Heraklion, Crete, Greece
E-mail: cgrivas@nhmc.uoc.gr
[2] Hellenic Centre for Marine Research – Crete, Thalassocosmos (Former U.S. Base at Gournes)
P.O. Box 2214, GR-71003 Heraklion, Crete, Greece
E-mail: kchris@her.hcmr.gr
[3] University of Crete – Natural History Museum of Crete
P.O. Box 2208, GR-71409 Heraklion, Crete, Greece
E-mail: sxirouch@nhmc.uoc.gr
[4] University of Crete – Natural History Museum of Crete
P.O. Box 2208, GR-71409 Heraklion, Crete, Greece
E-mail: mprobonas@nhmc.uoc.gr

Abstract

During the implementation of Project LIFE – NATURE 2002 "Conservation Actions for Bearded Vulture *Gypaetus barbatus* and Biodiversity in Crete" [LIFE02 NAT/GR/8492], Action D on recurring biotope management, included the intense and systematic monitoring of two active Bearded Vulture nests, with the use of a mini-camera and microphone system. The collection of detailed data on cainism, could lead – at a second stage – in the rescue of the second chick from a problematic nest and the establishment of breeding pairs in captivity, for the future restocking of the Cretan population. One of the systems was disassembled because of the pairs' inactivity and the monitoring project was continued with the other nest for three consecutive breeding periods. Except chick extraction, the monitoring system had several other targets, including the estimation of the exact period for supplement food provision, the proper time for marking/radio tagging the chick, provide early warning for any possible health issues (poisoning symptoms, thin eggshells, nest predation etc.), determine seasonal food availability (foraging sorties) and collect other breeding data (nest building, clutch size, nest relieves, food delivery rate, nest defense etc.), which could provide crucial information for the conservation of the species. Finally, some of the video material could be implemented in the public awareness campaign. To spare resources the systems' set up was towards the optimum possible automation. Evaluation of the system was based on the number of recording days during the breeding period. Furthermore, the non-recording days were analyzed according to technical causes, so to give an evaluation of their importance in the total lost time. These technical problems had to be solved before the system functioned adequately in the third year, although some of the flaws, e.g. power supply in the nest area, remained as a disadvantage of the general conception.

ANNEX II

ABSTRACTS OF POSTERS



USE OF CARCASSES FROM WOLF KILLS BY GRIFFON VULTURES IN EASTERN RHODOPES, BULGARIA

Ivaylo Angelov, Dimitar Demerdjiev & Stoycho Stoychev

Bulgarian Society for the Protection of Birds / BirdLife Bulgaria
Regional Office – Haskovo, P.O. Box 130, Haskovo 6300, Bulgaria
E-mail: haskovo@bspb.org

Abstract

BSPB/BirdLife Bulgaria conducted a two-year long survey (2004-05) on the conflict "predator-man", part of which was the investigation on the use of natural food by the Griffon Vultures.

Information for Griffon Vulture presence in 73 village territories was collected. In 44 of the territories the vultures have been seen feeding on a carcass, while in the rest 29 they have been recorded flying only. The data were collected trough observations and interviewing local people (shepherds, hunters, foresters), who know the species well, since in the last decades the area is traditionally inhabited by the vultures.

From August to November 2004 in an area of 2,450 square km with 159 villages, 157 domestic animals and at least 8 fallow deer, all of them killed by wolves, have been eaten by the vultures. The total number of domestic animals, killed by wolves is 365, which is very close to the real one, while according to the data from the Hunting Reserve "Studen Kladenetz" it is very diminished for the Fallow Deer. The quantity of the food, available for vultures is estimated through taking into account the species and the number of victims per attack.

The food for vultures, originating from wolf kills, exceeds the food provided and eaten by the vultures, during supplementary feeding: 6,500 to 3,900 kg. The population number of the Griffon Vultures, inhabiting the Bulgarian part of Eastern Rhodopes is 107-120 individuals.

Griffon Vultures are dependent to a significant extent on the natural food, definitive for their distribution. The core of the vulture population in Eastern Rhodopes inhabits the area in and around Hunting Station "St. Kladenetz". The Fallow Deer population and domestic animals, bred through an extensive way, here reach their highest density and bear the highest losses from wolf attacks in the Rhodopes, which provides a lot of carcasses and creates the most important area for Griffon Vultures in the Eastern Rhodopes.

DECLINE OF THE EGYPTIAN VULTURE *Neophron percnopterus* IN LA RIOJA (NORTHERN SPAIN)

Álvaro Camiña, Eugenio Montelío & David Martín

ACRENA
Apartado de Correos 339, 28220 Majadahonda (Madrid), Spain
E-mail: acamia@vodafone.es

Abstract

There is little information on the evolution of Egyptian Vulture *Neophron percnopterus* populations in mountainous areas. In La Rioja (Northern Spain) the species has experienced a severe decline from 35 pairs in 1985 (then up to 50 were estimated) to 27 in 1987, around 26 in 2001 and only 16 in 2005. However, the accuracy of censuses has been analysed in detail resulting in an overestimation of the population size. An average 5% yearly rate of decrease over this time has been recorded. Decreases with such a low rates can remain unnoticed for a long time. Monitoring of breeding parameters for the last three years showed that both productivity and fledgling success have fluctuated being normal for the species and similar to other Spanish populations. Productivity was 0.90 in 2004 and 1.11 in 2005 while the fledgling success 1.00 and 1.25 respectively. These parameters were similar for other Spanish populations. The population decline has been homogeneous in the whole region and irrespective of nests being located in Special Protected Areas or not. Some overestimates were clearly due to double counts of the same pairs. At least the 31.2% of pairs changed their nest location in the last three years. Change was made within the same cliff or other located in the vicinity. No clear reasons were found for this. Diet of the Egyptian Vulture in the area is difficult to analyse due to the landscape characteristics that biases results. The only prey remains showed the dependence from pig and rabbit intensive farming and the species has been recorded also at cattle raising areas both intensive and extensive. Thanks to individual identification based on plumage features for some pairs foraging area was estimated around 22 km radius around the nest. Poison is still present but not common so it could be a reason for the decline that is urgently to measure. An urgent conservation plan is required to stop the regressive trend of the Egyptian Vulture in the region.

THE DICLOFENAC: COULD A VULTURE CRISIS HAPPEN IN EUROPE?

Álvaro Camiña & Eugenio Montelío

ACRENA
Apartado de Correos 339, 28220 Majadahonda (Madrid), Spain
E-mail: acamia@vodafone.es

Abstract

A veterinary drug (Diclofenac) has been identified as the major cause of the vulture's decline in Asia. For decades, vultures in Spain have fed from carcasses illegally disposed at feeding sites called *muladares*. However, little information exists on the veterinary drugs and other products that are administered to livestock in relation to vulture conservation. Used leaflets, tins, bottles and bags left at these places were collated and listed. The 80% of the feeding sites visited provided with remains of veterinary medicines. Up to date a total of 20 different products have been identified. Among heavy metals lead and zinc were detected from intensive pig rearing practices. Both are used at the first stages of piglet's growth. Signs of contamination by those heavy metals have been found in both the Eurasian Griffon and Bearded Vulture respectively. Despite of lead could be also ingested from bullets used in hunting, vulture colonies or pairs feeding from certain intensive farming in their foraging areas could be exposed to such potential contamination. Identifying these kinds of intoxications is easier for endangered species such as the Bearded Vulture, due to a yearly-established monitoring programme. Nevertheless, is more difficult for the Eurasian Griffon, comprising in Spain the largest population in Europe and only monitored on a 10-year basis. Food quality and not only food availability could play an essential role in vulture management. Further research on this topic is needed.

FOOD SHORTAGES FOR THE EURASIAN GRIFFON VULTURE *Gyps fulvus* IN LOS MONEGROS (EBRO VALLEY, ARAGON REGION)

Álvaro Camiña & Eugenio Montelío

ACRENA
Apartado de Correos 339, 28220 Majadahonda (Madrid), Spain
E-mail: acamia@vodafone.es

Abstract

This poster investigates the effect the carcass removal programme against the Bovine Spongiform Encephalopathy (BSE) has on Griffon Vulture population. In the mid Ebro Valley food availability for Griffon Vultures has drastically declined within the last year. Livestock farms in the area were regularly monitored for carcass disposal and food availability before and after the removal programme was established. Out of 93 farms 70 reared pigs while the remaining included sheep/goats (11), cattle (11), chicken or rabbits (5) and horses (1). Griffon Vulture censuses were carried out at the same time as surveys of food availability. The diet was almost exclusively comprised of pigs (there were no other medium or large carcasses available). Food availability has decreased within the last year (from around 3 carcasses per farm and day in 2004 to 0-0.5 carcasses in 2005). Carcasses are not available for vultures but Griffon numbers using the area still remain in similar numbers. Most of the vultures previously present before were non-breeding birds (54.04%) that probably increased their survival by feeding far from the breeding colonies. After the carcass removal programme the situation has reversed, with adults outnumbering immature birds (42.91%). As a conclusion, the establishment of carcass removal programmes all over Spain can threaten the stability and future evolution of Griffon Vulture populations. Scientific research before any management measures is urgently needed for a proper conservation policy for vulture species.

GRIFFON VULTURE *Gyps fulvus* ARTIFICIAL REPRODUCTION

Savvas Iezekiel[1] & Haris Nicolaou[2]

[1] Cyprus Association for the Protection of Avifauna
10 Kalamatas street, 8047 Paphos, Cyprus
E-mail: iezekiel@cytanet.com.cy
[2] Cyprus Association of Professional Foresters, Cyprus Forestry Department
Louki Akrita street, 1414 Nicosia, Cyprus
E-mail: nicolaouharis@cytanet.com.cy

Abstract

One of the most threatened bird species in Cyprus is the Griffon Vulture, which thirty years ago used to be a fairly common species on the island.

The protection and conservation of rare raptor species such as the Griffon Vulture can be significantly supported by artificial reproduction. Eggs normally are lost by parental neglect, predation, extreme environmental conditions, pathogen infection and other calamities. Vultures lay a second clutch to replace eggs that are removed for artificial incubation so we can achieve double reproduction since vultures lay only one egg every year.

During the project for the conservation of the Griffon Vulture in Cyprus, many management measures/activities were implemented in order to achieve the project's aims and contribute towards conserving the indigenous vulture population.

Among these measures, a cage with the proper specifications was constructed to encourage breeding in captivity in the cage. An attempt was made successfully in 2004 for artificial reproduction of a vulture under laboratory conditions. The egg was removed from a pair in captivity and after it was incubated artificially it was placed in a nursery for a certain period and then in an artificial nest until the age of 4 months old. Then it was transferred back to the cage where its natural parents were found when it was ready to survive by itself without any human support.

We are planning to continue this effort in the following years with more eggs since 4 pairs are mature for reproduction in captivity as one of the measures to increase the population of the Griffon Vulture in Cyprus.

THE CENSUS OF VULTURES (AEGYPIINAE) IN HERZEGOVINA BEFORE THE BALKAN CIVIL CONFLICT

Saša Marinković[1], Ljilja Orlandić[2] & Branko Karađić[1]

[1] Institute for Biological Research, Despota Stefana 142, 11000 Belgrade, Serbia & Montenegro
E-mail: sasafond@ibiss.bg.ac.yu
[2] Birds of Prey Protection Fund, Despota Stefana 142, 11000 Belgrade, Serbia & Montenegro
E-mail: grifon@ibiss.bg.ac.yuz

Abstract

In 1992, due to the beginning of civil conflicts in the Balkans, Griffon Vulture populations from East Herzegovina moved towards surrounding territories, which were not directly influenced by military operations. Certain number of birds migrated to Serbia and inhabited the gorge Uvac, which was well documented by sharp increase of local population noted in 1995. The current state of Griffon Vulture local population in the gorge Uvac and its geographical position (located in northeast direction some 168 km far away) offer opportunities for spontaneous recolonisation of previously abandoned habitats in Herzegovina. This is supported by the fact that during winter months young birds from Serbia migrate through Herzegovina. On the other hand, during this phase of their life cycle, they are facing risk to be poisoned. Presented results could be used as a basis for planning protection and reintroduction of Griffon Vulture in Herzegovina.

Successful protection and reintroduction achievements of vulture species in Spain and France confirmed that it is possible to return these species on locations from which they have already vanished. By launching the Action Plan for vulture protection in the Balkans, Bosnia and Herzegovina got the opportunity to be included in reintroduction programmes of endangered vulture species.

A long-term study (1980-1991) has been performed, using census of nest and nesting couples. During this period, 61 nests, 83 nesting couples and 252 cases of nesting have been observed in four colonies of Griffon Vulture. During this period, 6 nests and 10 cases of nesting have been observed for the Egyptian Vulture. One pair of Bearded Vulture has been observed; however, the nest was not found.

AGE DETERMINATION OF BLACK VULTURE *Aegypius monachus* PULLI IN THE NEST

Javier de la Puente & Javier Gamonal

Área de Estudio y Seguimiento de Aves, SEO/BirdLife
Melquiades Biencinto 34, E-28053 Madrid, Spain
E-mail: carricero@worldonline.es

Abstract

It is often very useful to know the age of chicks in the nest. For instance, if activities such as ringing or attaching transmitters are to be carried out correctly, it is essential to know the age of the pulli before accessing the nest. If chicks are tagged at the wrong age there is the risk of unnecessary disturbance or even serious problems.

However, the monitoring of a Black Vulture colony comprises relatively few visits to the colony, resulting in it not being possible to accurately determine the laying or hatching dates for most of the clutches. Consequently, it is not possible to accurately age chicks in the nest. However, with some other suitable information it is possible to age chicks in the nest, even without knowledge of the laying/hatching date. With this information and knowledge of the average incubation period for the species, it would be possible to accurately estimate the age of pulli in the nest that can be seen well through a telescope. Knowledge of the breeding phenology of both each pair and the colony as a whole is very important for their management and conservation.

A number of illustrations and descriptions for accurately estimating the age of Black Vulture pulli in the nest with very few error days are presented in this study. Twelve development stages of the pullus, specifically 10-day periods, from birth until fledging approximately 120 days later, are described here.

Digiscoped photographs taken weekly at four nests of black vultures have been used, where the precise hatching date of the chick was known. Additionally, several hundred digital photographs, taken at the time of ringing of about 65 chicks of known age, have been used. The information used within this study was gathered during monitoring carried out at the Black Vulture colony at Rascafría (Madrid, Spain) in years 2004 and 2005, by Sociedad Española de Ornitología (SEO/BirdLife) for the “Consejería de Medio Ambiente y Ordenación del Territorio de la Comunidad de Madrid”.

DETECTION OF NON-STEROIDAL ANTI-INFLAMMATORY DRUGS (NSAIDs) IN THE HAIR OF LIVESTOCK ANIMALS AND IN THE FEATHERS OF SCAVENGING BIRDS OF PREY

N.L. Richards[1], K.S. Scott[1], S. Hall[1] & N. Harrison[2]

[1] Department of Forensic Science and Chemistry, Anglia Polytechnic University
East Road, Cambridge England CB1 1PT, United Kingdom
E-mail: ngaio_richards@hotmail.com
[2] Department of Life Sciences, Anglia Polytechnic University
East Road, Cambridge England CB1 1PT, United Kingdom

Abstract

Non-steroidal anti-inflammatory drugs (NSAIDs) are administered to livestock animals to alleviate lameness. The NSAID diclofenac was recently implicated in the near extinction of *Gyps bengalensis*, *G. indicus* and *G. tenuirostris* on the Indian subcontinent. These species were exposed to the drug when they consumed livestock carcasses. Other avian scavengers may also be secondarily exposed to NSAIDs wherever these are administered to livestock for veterinary purposes. Presence of NSAID residues in meat left at feeding stations or provided during rehabilitation efforts may also be of concern.

Drugs can be detected in hair long after they would cease to be traceable in tissues. Our objective is to develop forensic methodology that will enable the detection of NSAIDs in feathers and animal hair and could be used as an alternate diagnostic tool if avian carcasses are received with tissues too deteriorated for analysis, or to evaluate the safety of meat offered as food to scavengers. We are also interested in looking at residues in hair and feathers of individuals in relation to those found in the muscle, liver and kidney as a means of quantifying potential exposure to scavengers.

We welcome international collaborators and seek hair and feather samples from animals in the wild that may risk exposure, from animals administered NSAIDs as part of a treatment course, or from subjects being dosed with NSAIDs as part of a study. Examining secondary exposure to NSAIDs both in several avian scavenging species and in different agricultural scenarios also provides a unique opportunity to compare the relative tolerance of each species to NSAIDs and to explore their pervasiveness in the agricultural environment and food chain. From this research, we hope to develop a sampling and laboratory protocol that can be adapted by species of concern, by veterinary drug, by country and by agricultural context.

VULTURE PARADISE IN THE KATERNIAGHAT WILDLIFE SANCTUARY, UTTAR PRADESH, INDIA

S.M. Satheesan[1] & Khan Shamshad[2]

[1] B-404, Crystal Palace Rambaug, AS Marg Powai, Mumbai 400076, India
E-mail: smsatheesan2001@yahoo.com
[2] Flat 1, St. Clements' Court, 65 Cleveland road, Bournemouth BH1 4QJ, United Kingdom
E-mail: koolroma@yahoo.co.uk

Abstract

When the rest of the Indian sub-continent was facing a colossal vulture decline since 1990, the Katerniaghat Wildlife Sanctuary continued to be their paradise. This protected area spread over 400 sq. km harbours the Tiger, One-horned Rhinoceros, Elephant, and Leopard in India and bordering Nepal, as well as Crocodile, Gharial, and the Gangetic Dolphin in the Gerua river. In this Forest Division in Bahraich District 28º24' - 27º4' N to 81º65' - 81º3' E covering 551.64 sq. km, 575 vultures of five species (Long-billed and Eurasian Griffons, and White-backed, King, and Egyptian Vultures) and 31 nests of the Oriental White-backed Vulture *Pseudogyps bengalensis* were sighted in February 2002. But their actual population may be much more because of the proximity to Nepal and the Himalayas and the authors had sighted 25 White-backed and 300 Egyptian Vultures in August 2005. The reduced population of White-backs observed here resembles similar trends in population decline observed during the rains elsewhere. Moreover, vultures return to Katerniaghat to breed year after year, further confirmed by the vestiges of nesting materials detected on, and wing primaries found below *Semal* and *Haldu* trees. Authors unearthed a 'kaleidoscope' of factors responsible for vulture decline here, including man-animal and animal-animal conflicts and other threats. Diclofenac and other NSAIDs were and are not used on cattle here. A cloudburst of ideas leading to new conservation techniques to save vultures and other scavengers are discussed. Vultures here need immediate and total protection so that they can continue to "fire-wall" tigers on the prawl in sugarcane fields, crocodiles lurking in lotus ponds and other species against deadly pathogens and maintain the health of ecosystems network.

ERROR ASSESSMENT OF A TELEMETRY SYSTEM FOR EURASIAN BLACK VULTURE *Aegypius monachus* IN THE NATIONAL PARK OF DADIA-LEFKIMI-SOUFLI FOREST, GREECE

Stefan Schindler, Dimitris Vasilakis & Kostas Poirazidis

WWF Greece, Dadia Project
Dadia, GR-68400 Soufli, Greece
E-mail: ecodadia@otenet.gr

Abstract

The Eurasian Black Vulture *Aegypius monachus* is a globally near-threatened species and its breeding population in the National Park of Dadia-Lefkimi-Soufli Forest, Greece is the last remaining in the Balkan Peninsula and a central subject of conservation. In order to improve the knowledge of the range use of this population, a radio telemetry study was applied to estimate the home range as well as the movements and the activity patterns of the birds.

The applied system of telemetry was optimised by a study of error assessment, concerning the best estimation of the point locations and the determination of their precision. Data on the test transmitters were collected exactly like those for the telemetry study of the Eurasian Black Vulture. Transmitter stations consisted in a transmitter lifted in the air with a balloon, which was filled up with Helium.

We found out that the system was without directional bias, that the standard deviation, as measure of accuracy of the directional bearings, was 9.7°, and that the best performing location estimator was the Andrews estimator. Triangulations performed better than biangulations. Regarding the 156 successful location estimates obtained by triangulation, the average linear error between estimated and true locations was 1 km, while the 108 location estimates obtained by biangulation provide on average a linear error of 1.6 km. The median value of the area of the 95% confidence interval (CI) ellipses that we computed for the triangulation data set was 1,048 ha. The true location of the transmitter station was in 90.4% of the cases inside the 95% CI ellipse, thus we could prove that the ellipses are an accurate measure of the confidence interval. We conclude that the applied system of telemetry is precise enough to evaluate home ranges of vultures and to determine their main areas of foraging.

PRESENCE OF VULTURES AND OTHER CARRION EATING BIRDS IN THE ARTIFICIAL FEEDING SITE, ON MT PINOVO, ARIDEA, N. GREECE

L. Sidiropoulos, P. Konstantinou, P. Azmanis & R. Tsiakiris

Environmental Group of Aridea "Chrysaetos"
Village of Garefi, GR-58400 Aridea, Prefecture of Pella, Greece
E-mail: xrysaetos@hotmail.com

Abstract

We examined the utilization of an artificial feeding site (AFS) by carrion eating birds in Pinovo, a mountain chain near the northern Greek borders (highest peak 2,156 m.), which is one of the most important areas for raptor conservation in the Balkans. The AFS is located at 530 m. a.s.l., in a dry southern slope, nearby a small limestone gorge. The AFS was supplemented regularly, with slaughterhouse offal and carrion from autumn 2002 up until the autumn of 2005 and monitored thereafter for 2-4 days. Of the 31 species of birds of prey recorded in the area, 9 – including all four species of European vultures – were observed feeding, along with corvids. Raven was the most regularly observed species, followed by the Golden Eagle, especially in winter. The Egyptian Vulture was the most regular vulture species in the AFS during spring and summer months, while Griffon Vultures were abundant in autumn. Especially in autumn 2004, more than 20 Griffon Vultures concentrated and some remain in the area up until late November, because of the continued supplementing of the AFS. The Bearded Vulture seems to use the AFS mainly from late winter to early spring, coinciding with its regular presence in the area. Common Ravens may appear in large concentrations all year-round, especially on large amounts of slaughterhouse offal, but do not appear much on disposals of carrion. In 2004 and 2005 the AFS functioning was discontinued for some months due to various logistic reasons and this had a profound effect on the presence of several species in the wider area, especially concerning Bearded and Griffon Vultures. Thus, it is necessary to supplement and monitor the AFS on a regular basis, as part of a larger AFS network in the whole mountain range, in order to safeguard all declining species of carrion eating raptors.

VULTURES IN MODERN GREEK FOLK HISTORY AND LEGEND

Kalliopi Stara[1] & Rigas Tsiakiris[2]

[1] Department of History and Archaeology, University of Ioannina, Ioannina, Greece
E-mail: kallio21@hotmail.com
[2] Department of Ecology, School of Biology, Aristotle University of Thessaloniki, Thessaloniki, Greece
E-mail: rigast@hotmail.com

Abstract
Birds are not only part of our natural heritage but also important components of our cultural history. From the antiquity until recently, vultures are always present in Greek legends and traditions. In order to investigate people's knowledge about vultures we interviewed local people in Western Greece. Local author's texts and ethnographic material were also reviewed. Vulture names differ from place to place. For residents and transhumants of Western Greece, spring is coming with Egyptian Vultures, named "Cuckoo's horses", which carry migrating Cuckoos on their backs. Their arrival is linked with children's couplets and magic actions for health, fortune, marriage and success in dairy products. In other places Egyptian Vulture is called "cheese-maker". Moreover, the bird's body is used in folk medicine. Griffon Vultures are present in every day sayings, characterizing lazy, boorish or gluttonous people. Fairy tales personalize vultures and eagles as shepherds, while in local traditions sheep are transformed to Griffon Vultures due to the supernatural punishments of shepherd's inhospitable behaviour. It is known that vultures ate unburied people killed in wars, but legend says: "heroes were eaten by eagles"! Instruments of pastoral music tradition are also associated with vultures. Children used to collect flight feathers and sell them to local lute players, while flutes were frequently constructed from the ulna of the wing of Griffon Vultures or Golden Eagles. Nevertheless, those bones had to remain 40 Sundays in church before use, to be demons-purified. Lastly, we found many place names referring to vultures, but after their population decline, people rarely associate those toponyms with birds. Folk history and legend remain longer than birds themselves, but if we want to involve local people in nature conservation it is crucial to save local history, to investigate perceptions about endangered species and to create locally grounded environmental education and information material.

FWFF ACTIVITIES FOR REINTRODUCTION OF THE GRIFFON VULTURES AND CREATING SUITABLE CONDITIONS FOR THE CARRION EATING BIRDS IN BULGARIA

Emilian Stoynov

Fund for the Wild Flora & Fauna (FWFF)
2700 Blagoevgrad, P.O. Box 78, Bulgaria
E-mail: pirin@fwff.org

Abstract

The predator attacks over livestock is the most common reason for use of poisoned baits and related poisoning of wildlife (e.g. vultures). To solve this problem FWFF has established a compensation program for livestock losses caused by predators. The FWFF created its own compensation herd of sheep and goats and provides a live animals for each killed by predators to the local farmers in the target region.

Feasibility study for the re-introduction of the Griffon Vulture in Eastern Balkan and Rila Mountain were provided by FWFF. The results show that the re-introduction of the species in these areas is possible and even recommended as a bridge between the naturally survived populations and as a precondition for recovery of the other vulture species.

MONITORING OF THE EURASIAN GRIFFON VULTURE *Gyps fulvus* IN EASTERN SPAIN (CASTELLÓN PROVINCE)

Martí Surroca, Juan Jiménez & Alvaro Camiña

ACRENA
Apartado de Correos 339, 28220 Majadahonda (Madrid), Spain
E-mail: acamia@vodafone.es

Abstract

The Eurasian Griffon Vulture has experienced a great increase in Castellón province (Eastern Spain) over the last 30 years from around 3 breeding pairs in 1973 (1 breeding site) to 204 in 2004 (42 breeding sites). This populations distributes In order to mitigate a probably food shortage caused by the Bovine Spongiform Encephalopathy (BSE) the Conselleria from Territori I Habitatge from the Local Government of Generalitat Valenciana started a programme to preserve the carrion eating birds in that region. One vulture restaurant operates from February to June at Cinctorres Municipality. During 2005 a marking programme has been implemented. Griffon Vultures are being fitted with wing tags and PVC rings, which permits the individual identification. Both marks show the same alphanumeric combinations on a yellow background (numbers and/or letters). Griffons are captured with a trap, which is baited with pig carcasses. Capturing started in July 2005 and up to October, 25 birds have been caught. Samples of both feathers and blood are being collated for further sexing and heavy metal analyses. In a short time a sample of adult birds will be fitted with Radio-transmitters.

REINTRODUCTION OF THE GRIFFON VULTURE *Gyps fulvus* IN NEBRODI REGIONAL PARK, SICILY

Massimiliano Di Vittorio

Parco dei Nebrodi (Sicily)
Via Cosenz 155, 98076 S. Agata Militello (Me), Sicily, Italy
E-mail: elemass@neomedia.it

Abstract

The Nebrodi Regional Park is the biggest protected area in Sicily, containing 25% of the island's forests. The Griffon Vulture, until the mid-1900s, was common in Sicily (Benoit 1840, Minà Palumbo 1853, 1857, Doderlein 1869-74, Massa 1985, Iapichino & Massa 1989) and the last Sicilian population lived in the Nebrodi Regional Park until 1969. In this year, the Griffon Vulture became extinct in Sicily due to a poisoning event (Priolo 1967). A reintroduction plan for this species has been underway in the Nebrodi Regional Park since 1998, with the main aim of increasing biodiversity in this conservation area. As a scavenging species, Griffon Vultures also fulfil an important ecological role. With the reintroduction of nearly 40 vultures during the period 1998 to 2004, a small population of 9 Griffon established itself. In December 2004 the vultures initiated breeding and in February 2005 3 eggs were laid by two pairs. This is an exceptional event, because usually the Griffon vulture laid just one egg, and not seem the case of two female that use a same nest (Brichetti *et al*. 1992; Cramp & Simmons 1980). During the last week of April 2005, three Griffon Vulture eggs hatched, and in the second week of august the young vultures has fledged, marking the first Sicilian breeding of this species in more than 40 years. To promote the success of this project, the Park administration has recently established two feeding stations and these are already been used by the small vulture population. To continue with the reintroduction programme in the Nebrodi Regional Park, others 6 Griffon Vultures has been released in September and now the population is composed of 17 specimens and are in course the import of additional 50 birds from Spain.

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NEST AND NEST TREE CHARACTERISTICS OF CINEREOUS VULTURE *Aegypius monachus* IN THE TURKMENBABA MOUNTAIN, NORTHWEST TURKEY

Elif Yamac[1], C. Can Bilgin[2] & A. Yavuz Kilic[1]

[1] Department of Biology, Anadolu University
26470, Eskisehir, Turkey
E-mail: eerdogdu@anadolu.edu.tr
[2] Department of Biology
Middle East Technical University, Ankara, Turkey

Abstract

Cinereous Vulture *Aegypius monachus* is a globally threatened raptor and for the conservation of species, it is very important to determine the nest properties and nest tree characteristics. After Spain, Turkey has the second largest population in Europe. In this study, nest structure was detected and nest-tree characteristics evaluated in the Türkmenbaba Mountain, Eskisehir (northwestern Turkey), where the largest *Aegypius monachus* colony in the country exists. Individual nest and nest trees preferences were identified throughout 2001-2002. A total of 73 nests were recorded in 17,500 hectares. The diameter axis of nest, diameter axis of nest cup and nest thickness as nest properties and tree species, height of nest tree, diameter at breast height (DBH), aspect of tree, trunk shape, nest-tree branches, the conditions of nest tree and crown class of nest tree as nest tree characteristics were examined. Any preferences for nest trees were statistically tested against randomly selected trees.

The nest structure measurements indicated that the mean nest diameter was 176.9 ± 42.63 cm, nest thickness was 37.13 ± 15.05 cm. and cup diameter was 62.11 ± 10.49 cm. The species invariably nests on flat-topped *Pinus nigra* trees with a height of 11.47 ± 3.87 m. (mean ± SE) and DBH of 42.91 ± 7.36 cm. Statistical tests for the nest trees revealed a significant preference to northeasterly-easterly directions. It was found that the trunk shape of the nest tree is not important statistically. *Aegypius monachus* showed a preference for building their nest on trees containing <20 branches per trunk and intermediate or upper level of canopy. According to our results, *Aegypius monachus* prefers older and mature *Pinus nigra* trees in Türkmenbaba Mountain. Therefore, the preservation of these kind of trees is essential for the survival of the species and should be incorporated into management plans.

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