Chapter 19
Old World Vultures in a Changing Environment

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Introduction

Carrion is a pulsed food resource of unpredictable occurrence in space and time that offers a high nutritive biomass, but is not globally widespread across all habitats and territories, and can be considered free because it does not require a large physical investment derived from predation (Ostfeld and Keesing 2000; DeVault et al. 2003; Selva and Fortuna 2007; Barton et al. 2013; Moleón et al. 2014a). As a result of these features, organisms feeding on this resource have developed morphological and behavioural adaptations to optimise its exploitation (Hertel 1994; DeVault et al. 2003; Moreno-Opo et al. 2015a, 2016), establishing guilds of species as in the case of carrion-eating birds (Selva and Fortuna 2007). Among terrestrial vertebrates, only vultures (families Accipitridae and Cathartidae) have evolved into obligate scavengers (DeVault et al. 2003; Beasley et al. 2015; Moleón et al. 2014b). Scavengers exploit carrion at different levels of intensity, and depending on their degree of carrion consumption and their own adaptive traits, they are defined as either obligate or facultative (DeVault et al. 2003; Wilson and Wolkovich 2011). As a result of the unpredictability in the availability of carcasses, the proportion of obligate scavenger species are scarce in comparison with facultative species that

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scavenge at variable rates but that can subsist on other food resources in the absence of carrion (DeVault et al. 2003; Beasley et al. 2015; Mateo-Tomás et al. 2015).

Worldwide there are 23 obligate scavenger species. Of the 16 Old World vultures, 81% are globally threatened or near-threatened. Most of these species are declining, particularly in Asia and Africa, mainly as a consequence of anthropogenic activities such as the illegal use of poisons, landscape transformation, health policies and ingestion of toxic veterinary drugs (Ogada et al. 2012a). These threats persist and continue to increase, despite vultures' importance to mankind. Vultures have traditionally provided important ecosystem services such as controlling diseases and pests, recycling nutrients and providing cultural inspiration and recreational value (Moleón et al. 2014a).

Vulture declines have far-reaching consequences, especially when considering the ecosystem services they provide to humans. It has been estimated that Spanish vultures remove >8000 tons of livestock carcasses per year, which prevents the release of greenhouse gases and provides economic savings estimated at 1.5 million Euros (Margalida and Colomer 2012; Morales-Reyes et al. 2015). Similarly, in developing countries such as India and Kenya, vultures likely play an outsized role in limiting disease transmission at carcasses (Markandya et al. 2008; Ogada et al. 2012b). In India, vulture declines have been linked to a subsequent increase in feral dogs, resulting in a rise in rabies transmission with an estimated $34 billion in healthcare costs between 1993 and 2006 (Markandya et al. 2008).

Vultures have biological adaptations that greatly complicate their conservation. Most significant among these is their need to range over vast landscapes in order to obtain food, which necessitates that conservation efforts be initiated across international boundaries (Margalida et al. 2013a, b; Lambertucci et al. 2014). In addition, they have evolved a life history strategy that emphasises long life and slow reproduction. Individuals do not begin breeding until *c*. 5 years of age, most species lay a single egg, and one breeding cycle can last up to 2 years in some species (see reviews in Mundy et al. 1992; Donázar 1993; Houston 2001). This ensures that once populations decline, they will take a long time to recover and the financial resources needed to restore depleted populations will be significant.

Perhaps not surprisingly, the conservation of Old World vultures varies widely by region. The European region is home to four vulture species of which three are threatened or near-threatened. Vultures were extirpated from several European countries by human activities in the nineteenth and twentieth centuries, but Eurasian griffon (*Gyps fulvus*) and cinereous vultures (*Aegypius monachus*) have been successfully reintroduced in recent decades to France; and bearded vultures (*Gypaetus barbatus*) have been reintroduced to Austria, Switzerland and Italy. Since 1993, the EU and various national governments have invested significant financial resources in the conservation of vultures – including at least 76 LIFE projects related to these species – and between 1993 and 2014 spent € 121.9 million, of which € 59.7 million came from European funds. During this period, Spain, home to 90% of all European vultures, invested € 72.8 million (€ 30.8 million received from the EU) on 38 projects related to vulture conservation (Margalida and Oliva-Vidal 2017).

In Asia, conservation efforts have become critically important after the Asian vulture crisis of the 1990s. Currently, Saving Asia's Vultures from Extinction (SAVE) coordinates a consortium of regional and international organisations that work to conserve and to advocate and fundraise for activities that sustain South Asia's vultures (www.save-vultures.org). Conservation activities include an intensive (and expensive) captive breeding programme.

In Africa, four species of vulture have recently been uplisted to critically endangered (hooded *Necrosyrtes monachus*, white-headed *Trigonoceps occipitalis*, white-backed *Gyps africanus* and Rüppell's *Gyps rueppelli*) and two species to endangered (Cape *Gyps coprotheres* and lappet-faced *Torgos tracheliotus*) as a result of severe continental declines (Ogada et al. 2016a; BirdLife International 2017). This has triggered new interest (and funding) for studying and monitoring populations (e.g. in Ethiopia, Ghana, Guinea Bissau, Mozambique, Zambia and Burkina Faso); despite this, the majority of African countries still lack even the most basic information on their vulture populations. Across all regions, conservation efforts are still menaced by anthropogenic activities that are jeopardising the viability of many species.

Here we describe several global or regional threats that, taken together, provide an overview of the main factors affecting the conservation of Old World vultures and, by extension, the important ecosystem services these species provide. We give examples of conservation measures that have helped to improve their management. Finally, we suggest what is required to ensure the long-term survival of Old World vultures.

Diclofenac, the Asian Crisis and Lessons Not Learned

In the 1980s, three *Gyps* vulture species endemic to South Asia were the most abundant large raptors in the world, but during the 1990s, their populations were reduced by more than 99% (Prakash 1999; Prakash et al. 2003, 2007) and all are now listed as critically endangered (BirdLife International 2017). The non-steroidal anti-inflammatory drug (NSAID) diclofenac was identified in 2004 as the primary cause of the rapid declines in Pakistan, India and Nepal (Oaks et al. 2004). Low-cost veterinary diclofenac-based products were being widely administered to livestock, some of which subsequently died, and due to cultural practices, their carcasses were widely available to scavengers. Sufficient residues remained in the carcasses of treated animals to cause acute renal failure and rapid death of vultures feeding on them (Taggart et al. 2007). Controlled experiments subsequently demonstrated that many species of *Gyps* vultures are susceptible to diclofenac poisoning including Oriental white-backed (*Gyps bengalensis*), African white-backed, Cape griffon and Eurasian griffon vultures (Oaks et al. 2004; Swan et al. 2006a, b; Naidoo et al. 2009). Since death rates of vultures are low under normal conditions, additional mortalities caused by veterinary drugs can have a profound impact on populations. Models indicated that contamination of just 0.3–0.7% of ungulate carcasses containing lethal levels of diclofenac was sufficient to cause vulture populations to decline at ~50% per year, as observed for one species in India and Pakistan (Green et al. 2004).

The Government of India enacted a ban on the production, importation and sale of veterinary diclofenac products in 2006. Similar measures were quickly taken in Pakistan and Nepal and then in Bangladesh in 2010 (www.save-vultures.org). This action was further supported by the identification of meloxicam as a suitable alternative drug that has proven safe for *Gyps* vultures (Swan et al. 2006a, b; Swarup et al. 2007; Naidoo et al. 2008). Since 2010, diclofenac prevalence in dead cattle has reduced by 70% (Cuthbert et al. 2014), and population declines in South Asia have slowed and may have reversed in some areas (Prakash et al. 2012; Chaudrhy et al. 2012). However, current usage is still sufficient to eradicate vulture populations (Green et al. 2006), and this has been attributed to human-use diclofenac, available in larger 30 ml vials, which makes it very convenient and cheap for its illegal use as a veterinary dose (Bowden 2015). These large 'multi-dose' vials for human use were finally banned by the Indian government in July 2015 (Bowden 2015). In the meantime, a number of other NSAIDs (e.g. carprofen, flunixin and ketoprofen) have been shown to cause mortality in *Gyps* vultures (Cuthbert et al. 2007), although the likelihood of these causing widespread mortality varies (Cuthbert et al. 2014); it illustrates the potential minefield among insufficiently tested pharmaceutical drugs and their cost to susceptible populations.

Despite this history, the government of Spain authorised the marketing of diclofenac as a veterinary pharmaceutical for use in cattle, pigs and horses in 2013 (Margalida et al. 2014b). Spain is important for the global conservation of avian scavengers, as it holds >90% of the European population of vultures, the entire population of the globally threatened Spanish imperial eagle (*Aquila adalberti*) and important numbers of red kites (*Milvus milvus*), which may also be at risk. Spain has a history of providing carcasses for wild scavengers to aid in their conservation. Spanish law (RD 1632/2011) facilitates this by allowing carcasses of farm animals to be left in the field in some protected areas or to be taken to 'muladares' (vulture feeding stations) to provide food for wildlife, including vultures (Margalida et al. 2012).

By law, diclofenac should only be administered under veterinary supervision and should not to be given to animals that are likely to enter the natural food chain. However, there remains a real risk that carcasses containing diclofenac residues will be consumed by vultures. Spain has an important livestock industry, with around 25 million pigs and 5.7 million cattle, and diclofenac is licensed for use in many clinical conditions that occur in these animals (Margalida et al. 2014d). Several studies have demonstrated that Eurasian griffon vultures are indeed accessing antibiotic residues in livestock carcasses provided at feeding stations (Casas-Díaz et al. 2016), and modelling of a simulated vulture population predicted that the potential rate of decline of the Spanish population of Eurasian griffon vultures due to diclofenac would be 0.9–7.7% per year (Green et al. 2016).

It is clear, then, that the risk to avian scavengers from veterinary use of diclofenac and other NSAIDs has not yet been evaluated adequately, and given the current global status of vultures, the precautionary principle ought to be evoked to ban diclofenac for veterinary use in the EU and other vulture range states, as has already occurred in South Africa, Iran and several Asian countries.

Additional trials evaluating the long-term impact of meloxicam and other veterinary drugs on scavengers, as well as reducing the availability of diclofenac in livestock carcasses, are needed to ensure the recovery of critically endangered *Gyps* vultures. In addition, veterinary treatment should be applied cautiously to reduce the prevalence of other anti-inflammatories, antibiotics and antiparasitic agents that currently contaminate carcasses (Donázar et al. 2009a; Casas-Díaz et al. 2016; Blanco et al. 2017a, b). Livestock management has important implications for the ecosystem services provided by scavengers (Dupont et al. 2012; Moleón et al. 2014a), as well as for the associated economic, human health and environmental effects (Margalida and Colomer 2012; Morales-Reyes et al. 2015, 2017).

Poisoning and Environmental Contaminants

Among raptors, vultures are particularly vulnerable to contaminants due to their reliance on carrion (Chaps. 10 and 11 in this book). As a result of their feeding habits, large numbers of individuals can be poisoned at a single carcass. In most cases, illegal poisoned baits are used to attract carnivores, provoking collateral mortality of facultative and obligate scavengers (Hernández and Margalida 2008, 2009a; Ogada et al. 2012a; Margalida 2012; Mateo-Tomás et al. 2012; Richards 2012). Although the use of poisons to manage carnivore populations has been banned in many countries, poisoning continues to be a common illegal tool used in all regions to manage game species and/or to protect livestock (Hernández and Margalida 2008, 2009a; Ogada et al. 2012a; Tingay et al. 2012; Berny et al. 2015; Bowden 2015). Accordingly, intentional poisoning of carnivores by humans is likely the most widespread cause of vulture mortality worldwide (Donázar 1993; Margalida 2012; Ogada et al. 2012a). For example, in Spain 8000 cases of illegal poisoning were reported during the period 1990–2010 with victims including 53 bearded vultures, 366 Egyptian vultures (*Neophron percnopterus*), 759 cinereous vultures and 2877 Eurasian griffon vultures (Margalida 2012).

The majority of poisons used are highly toxic pesticides. Many are carbamate pesticides that kill rapidly, and their detection post-mortem can be difficult, particularly in developing countries where preservation of suspected poisoned carcasses and access to expensive laboratory testing equipment are difficult (Ogada 2014). The most notorious wildlife poisons include carbofuran (trade name Furadan) and aldicarb (trade name Temik), and even strychnine is still used (Guitart et al. 2010; Ogada 2014; Botha et al. 2015). Although many of these pesticides have been banned particularly in the EU, their continued misuse to poison wildlife persists (e.g. Berny et al. 2015; Ruiz-Suárez et al. 2015), illustrating the inherent difficulties involved in regulating pesticides.

Widespread availability, low cost, ease of use and effectiveness have ensured that pesticide poisoning remains the generalized method used to predators and by association vultures. Worryingly, there is a growing use of poisons to intentionally harvest vultures, particularly in Africa (Ogada 2014). Vultures are routinely harvested for their body parts using pesticides, which are then sold for fetish, throughout West and Central Africa and in southern Africa (Beilis and Esterhuizen 2005; Mander et al. 2007; Saidu and Buij 2013; Buij et al. 2016). This method of harvest represents obvious, but as yet untested, potential hazards for consumers who may be ingesting or inhaling powders procured from vulture parts (Mander et al. 2007). Since 2012, there has been a dramatic rise in the intentional poisoning of vultures at the scene of poaching incidents, mostly involving elephants. Vultures are targeted by poachers because they rapidly discover poaching incidents where their overhead circling alerts the authorities to the location of these illicit activities (Ogada et al. 2016b).

Given the long-standing and global use of pesticides and other synthetic poisons to illicitly kill predators and other wildlife, innovative solutions are clearly paramount. In southern Spain, detection dogs have been trained to recover carcasses of poisoned wildlife; combined with state-of-the-art forensic analysis capabilities, these efforts have been successful in prosecuting wildlife poisoners (Fajardo et al. 2012). The detection dogs and their handlers also undertake regular patrols in known or suspected poisoning hotspot areas that alongside the prosecution of poisoners have been proven effective in decreasing and deterring vulture poisoning incidents in Spain (Fajardo et al. 2012). In southern Africa and in Kenya, poison response trainings have empowered rangers, police officers and community group members to recognise and rapidly respond to poisoning incidents. While the underlying causes of vulture poisonings still persist, these trainings have been shown to be effective in reducing the probability of extinction for critically endangered vultures (Murn and Botha 2017) and even in preventing poisoning incidents (Ogada pers. observ). It is clear that eliminating illegal wildlife poisoning is a complex task that must involve legal, educational, economic and punitive measures.

Ingestion of pellets or fragments from lead bullets poses another significant threat to scavengers (Hunt et al. 2006; Kelly et al. 2011; Lambertucci et al. 2011). Upon impact, lead bullets often fragment and become lodged in muscle and soft tissue where they become available to scavengers that consume viscera or muscle tissue from field-processed and unrecovered big game. Sublethal exposure to heavy metals may affect bone mineralisation (Gangoso et al. 2009), reduce muscle and fat concentrations (Carpenter et al. 2003), cause organ damage and internal lesions (Pattee et al. 1981) and reduce hatching success (Steidl et al. 1991). Lead exposure linked to hunting has been documented for threatened species such as Egyptian vultures (Gangoso et al. 2009) and bearded vultures (Hernández and Margalida 2009b; Berny et al. 2015), and it has been suspected as a source of contamination in endangered *Gyps* vultures in Africa (Naidoo et al. 2017).

Given the ongoing and perilous situation involving lead poisoning of California condors (*Gymnogyps californianus*) in the New World, legislation to minimise exposure to lead should be urgently implemented or improved, depending on the country, to reduce its effects on large predatory and scavenging birds.

Bovine Spongiform Encephalopathy in Europe

In 2001, a new European health crisis emerged with an outbreak of the so-called 'mad cow' disease. Acquired from cattle infected by bovine spongiform encephalopathy (BSE), it affects the brain and nervous system of humans and animals and leads to the degeneration of brain tissue, giving it a sponge-like appearance (EFSA 2017). The source of the disease is cattle feed prepared from BSE-infected animal tissues, such as the brain and spinal cord (EFSA 2017).

The outbreak led to the passing of sanitary legislation by the European Union (Regulation EC 1774/2002) that greatly restricted the use of animal by-products not intended for human consumption. Henceforth, all carcasses of domestic animals had to be collected from farms and transformed or destroyed at authorised facilities; as a result, many vulture feeding stations associated with stock farms closed down almost overnight. In Spain, it was estimated that 80% of all sheep and goat carcasses and 100% of all cow carcasses were being disposed of as per the new law (Donázar et al. 2009a, b; Margalida et al. 2010). Between 2005 and 2008 in the Navarre and Aragon regions of northern Spain, it is believed that 80% of all feeding stations were forced to close (Cortés-Avizanda et al. 2010; Margalida et al. 2014c).

The economic impact was also significant, as businesses had to invest in the collection of animal carcasses that were once destined to be consumed by vultures and other wild scavengers. In Spain, the cost of collecting a carcass was estimated at around 20 € per animal (Donázar and Margalida 2009; Donázar et al. 2009a), and a similar figure was estimated in France (Boumellasa 2004). Additionally, the industrial transformation of carcasses costs between 66 and 96 € per tonne (burning and recycling of animal remains) (Boumellasa 2004; Donázar et al. 2009a). Further environmental damage included the energetic costs of these industrial activities, for example, in the form of emission of greenhouse gases (Morales-Reyes et al. 2015).

As the collection of dead cattle became more widespread during the last decade, the behavioural and demographic effects on vultures appeared suddenly. For example, reports of vultures (mostly griffon vultures) attacking and killing cattle increased exponentially (Margalida et al. 2011a, b, 2014a).

The new regulations provoked an annual decrease in breeding success, reduced population growth, increased mortality of young age classes, changes in the diet and behaviour of Eurasian griffon vultures and a reduction on egg quality (Donázar et al. 2009a, b, 2010; Zuberogoitia et al. 2010; Margalida et al. 2011a, 2014a, c; Hernández et al. 2018).

It was patent that the restrictions imposed by this new EU legislation, which deprived scavenger populations of essential resources, flagrantly contradicted member states' obligations to conserve these threatened species (Tella 2001; Donázar et al. 2009b; Margalida et al. 2010). Fortunately, recommendations made by scientists, conservationists and conservation managers recently led to new EU guidelines allowing farmers to abandon dead animals in the field and/or at feeding stations (Margalida et al. 2012). This example illustrates how scientific arguments can trigger positive political action and help to reconcile conservation challenges and human activities (Sutherland et al. 2004; Margalida et al. 2012).

Collision and Electrocution Due to Energy Infrastructure

The worldwide increase in the production and development of both traditional and alternative power sources (Northrup and Wittenmeyer 2012) constitutes a serious and growing threat to avian scavengers (Chaps. 12 and 13 in this book). Among the most important risks, mortality from electrocution and collisions with power lines and wind farms has been documented for numerous raptor species (Ledger and Annegarn 1981; González et al. 2007; Margalida et al. 2008; Lehman et al. 2010; Guil et al. 2011, 2015), and collision with wind turbines is now the main cause of death for griffon vultures in Spain (see Tellería 2009; Carrete et al. 2009). Impacts of this mortality may be severe enough to affect long-term population size (Bevanger 1998) and population dynamics (Ledger and Annegarn 1980; Leshem 1985; Nikolaus 1984; Angelov et al. 2013). What is clear is that the energy sector represents an increasing threat to vultures (and other birds and bats) due to its rapid, global growth, with Asia, led by China and India, having the most cumulative installations (GWEC 2015).

In Europe, among all species studied, vultures are among those most frequently killed by collision with spinning turbine blades (Carrete et al. 2012). Even slight increases in mortality due to poorly placed wind turbines can significantly affect populations and potentially accelerate extinction of sensitive or endangered species such as the Egyptian vulture (Carrete et al. 2009). Wind energy infrastructure in Africa may jeopardise the existence of local or regional populations of several species (Drewitt and Langston 2006; Jenkins et al. 2010; Bellebaum et al. 2013). In southern Africa, where bearded vultures are now critically endangered in the region, with only about 100 pairs remaining, two wind farms are planned for Lesotho within this species' core habitat (Reid et al. 2015).

In Europe, effective guidelines for wind farm establishment have been carried out to place them far from sensitive species (Carrete et al. 2012), while in Africa, apart from South Africa, the development of national regulations and guidelines in regard to the establishment of wind farms lags far behind the current pace of development (Reid et al. 2015; Ogada pers. obs.). In addition, national or regional sensitivity mapping to guide the placement of wind farms is urgently needed throughout most of Africa and Asia. Sensitivity maps that are easily accessible to developers are a critical component for guiding the placement of wind turbines to avoid migratory flyways and other sensitive areas for vultures and other birds.

Frequently overlooked is the substantial threat posed to vultures and other large birds, particularly in developing regions, from the installation of hundreds of thousands of kilometres of new power lines that are necessary to transport power from these often remote wind-generating facilities to consumers.

Mitigation measures for reducing electrocution and collisions with power lines include reviewing the placement of new lines, removing the earth wire, fitting lines with markers or changing the pylon design. These have been developed in several countries, particularly in Europe, North America and South Africa (Lehman et al. 2007). In Spain, several efforts have identified mortality hotspots and conducted pylon and line modifications (Tintó et al. 2010; Guil et al. 2011). In South Africa, collision rates of some species such as cranes and bustards were partially reduced after attaching bird-flight diverters on ground wires (Anderson 2002). Regrettably, the actual effectiveness of these measures for vultures is unknown, although for other raptor species some reductions in mortality have been reported.

The Future of Old World Vultures in a Changing World

Populations of obligate scavengers have declined significantly over the last several decades across the globe, mainly due to a suite of anthropogenic factors. In particular, habitat loss and human persecution have played a prominent role in vulture declines in many regions; however, unintentional poisoning has emerged as one of the greatest threats to avian scavengers globally (Oaks et al. 2004; Ogada et al. 2012a).

At the same time, the conservation needs of vultures represent a uniquely complex mix of factors. Apart from the often confounding influences of economics, politics, conservation science, human health, industrial development and veterinary practices as illustrated here, vultures' biology and life history strategy further complicates their management and conservation. For example, few efficient supranational management strategies exist, which inevitably limits the effectiveness of any conservation strategy (see Margalida et al. 2013a, b).

In order to reverse the negative trend for Old World vultures, innovative and collaborative management solutions are needed. For example, the Endangered Wildlife Trust in South Africa has developed a strategic partnership with their national power provider, Eskom, to minimise the negative impacts associated with electrical infrastructure on vultures and other wildlife through risk and impact assessments, training and research (www.ewt.org.za). More such partnerships will be needed with the energy sector, particularly in developing regions where limited capacity and access to resources favour such collaborations.

Other possible interventions include the creation of large feeding stations, which may involve the private sector (as suppliers) and the public sector (as managers). While feeding stations are an attractive solution from a management perspective, they must also be carefully evaluated in terms of their conservation effectiveness and appropriateness based on local conditions (Cortés-Avizanda et al. 2010, 2016; Moreno-Opo et al. 2015a, b).

Cultural services provided by avian scavengers offer a growing economic benefit associated with vulture and eagle viewing and photography, which is an emerging and powerful conservation tool, provided that a part of the profits are allocated to avian scavenger management programmes and also benefit local communities (Becker et al. 2005; Piper 2005; Donázar et al. 2009a).

Throughout the regions inhabited by Old World vultures, the challenges to their conservation are shaped by local conditions and cultural practices. Similarly, any conservation solutions must inevitably involve local people and communities who share the same landscapes and benefit from the ecological services provided by vultures.

The future of Old World vultures depends on developing long-term, collaborative research and conservation strategies that recognise the economic, political, social and cultural realities inherent in the various regions where these birds persist.

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