

Chapter 2

Mammal Species Extinction and Decline: Some Current and Past Case Studies of the Detrimental Influence of Man

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Introduction

The majority of animal species in the world are significantly and negatively influenced by human activities. The effects of human activity extend to all continents and from the high mountains to the bottom of the ocean. Regarding mammals, the latest data from the International Union for Conservation of Nature (IUCN) show that 1.4 % of assessed species are already extinct (EX) or Extinct in the Wild (EW) and a further 21 % belong to threatened categories, that is, Critically Endangered (CR), Endangered (EN) or Vulnerable (VU). Species in these categories thus comprise almost one-quarter of the 5513 mammalian species assessed by the IUCN (IUCN Red List version 2014.3). As it may take many years to demonstrate that a species really is extinct, the number of species listed as extinct in the Red List may be lower than in reality (Hilton-Taylor et al. 2009). Moreover, due to insufficient information 14 % of known species were categorised as ‘Data Deficient’ (IUCN 2014) so the proportion of threatened species may be even higher.

Although extinctions are a part of evolution, anthropogenic environmental change has greatly accelerated the rate at which extinctions occur, especially in the past few centuries (Sodhi et al. 2009). At present naturally caused extinctions are

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thought to be an irrelevantly small fraction of the total of modern extinctions (Hogan et al. 2010).

In this chapter we concentrate on terrestrial mammals and present case studies of species that have been categorised in the past or are at present extinct (EX), at risk of extinction (EW or CR), or declining (EN and with a decreasing population trend), and that were or are negatively influenced by habitat loss, fragmentation or degradation—the most frequent anthropogenic threats.

Causes and Patterns of Decline and Extinction

Causes

Any phenomenon that can cause mortality rates to exceed reproductive replacement over a sustained period can cause a species to become extinct (Sodhi et al. 2009). Small populations are more susceptible to extinction because of demographic and environmental variability (Pullin 2002), and also because of reduced gene exchange which, for example, decreases the likelihood of surviving a novel pathogen.

The most common anthropogenic threats are summarised in Table 2.1 along with some examples of species suffering from them. Habitat loss, fragmentation and degradation seem to be the most frequent threats to wildlife. These threats result from continuous human population growth, intensification of agriculture and forestry, and development of industry. Habitat loss and destruction are often exacerbated by other anthropogenic pressures such as uncontrolled hunting or poaching, introduction of new species, and war and civil unrest (Table 2.1), with climate change also now included in the list of threats. Eighty-one percent of terrestrial mammal species with an EN/CR/EW/EX classification are threatened by more than one of the 11 broad categories of threat used by the IUCN, with 10 % threatened by five or more, according to our analysis of the IUCN Red List (version 2014.3).

It seems that deforestation is currently and probably will remain the principal direct and indirect cause of local extirpations of species (Sodhi et al. 2009) as forests are the broad habitat with the highest mammal species richness, and also the highest number of threatened mammal species (around one-fifth of all threatened species). Legal logging or illegal timber extraction and forest destruction may be aimed at acquiring wood or at preparing areas for settlements and agriculture. Over a third of the tropical forest biome is now covered by croplands or herbaceous plant cover (Mace et al. 2005) and deforestation remains the most frequently recorded form of land-use change (Lepers et al. 2005). Deforestation by region from 2000 to 2010 was highest in Africa and South America (0.56 and 0.46 % lost per year, FAO 2010), with the greatest loss by area in Brazil and Indonesia. South-east Asia in particular has an unfortunate combination of deforestation, many islands, and high population; many of our example species in Table 2.1 and in the case studies are

from the region. Up to 21 % of south-east Asian forest species are predicted to be lost by 2100 because of past and ongoing deforestation (Sodhi et al. 2009).

Besides deforestation, agriculture is an important driver of local (and therefore potentially global) species extinction. Some of these effects are related to habitat availability and quality. For example, the development or intensification of agriculture and animal breeding (e.g. cattle ranching) are connected with loss of habitat resulting from drainage, cultivation, and use of fertilisers and pesticides (e.g. Gibbs et al. 2009). However, agricultural abandonment may also result in unfavourable changes to habitat. For example, land abandonment and lack of cattle grazing in some areas within the range of the saiga *Saiga tatarica* resulted in a decline in grassy species and encroachment of species inedible for the saiga (Larionov et al. 2008). In addition there are agricultural effects not related to habitat alteration; for example, domestic animals may transmit diseases or parasites, or compete with wild species for food. Other threats to habitat are the development of industry (e.g. salt extraction or various branches of mining) and the expansion of human settlements and associated infrastructure such as roads or dams, which leads to the isolation of small populations and restriction of migration routes.

Human activities cause direct declines in species numbers through uncontrolled hunting, poaching or persecution but also because of warfare, capture of animals as pets and trade in animal body parts. The IUCN Red List assessments cite biological resource harvesting as a threat to almost a quarter of threatened terrestrial mammal species (IUCN Red List version 2014.3). Direct conflicts with people often result in the killing of animals as a reaction to crop depredation or livestock predation by feral carnivores, both of which are frequently an outcome of earlier habitat encroachment by humans. Human hunting activities may also influence animal species indirectly, for example, by exploitation of the prey of a given carnivore. Exotic species intentionally or unintentionally introduced by people may become serious competitors or predators for indigenous species; this threat includes also transmitting new diseases or parasites against which autochthonic species may be unable to defend themselves (Sodhi et al. 2009). In particular, the rates of extinctions occurring on islands have been considerably increased by the introduction of novel predators (Sodhi et al. 2009).

Geographic Patterns

The regions with the highest overall density of mammal species are undoubtedly tropical: Mesoamerica, tropical South America, Sub-Saharan tropical Africa, south and south-east Asia. Half of the top 20 countries for numbers of threatened species are in Asia (Vié et al. 2009); the areas of highest total number of threatened mammal species are Indonesia (184 species), Mexico (100), India, Brazil and China in ranked order (Vié et al. 2009). The ranges of many threatened species are too small to be

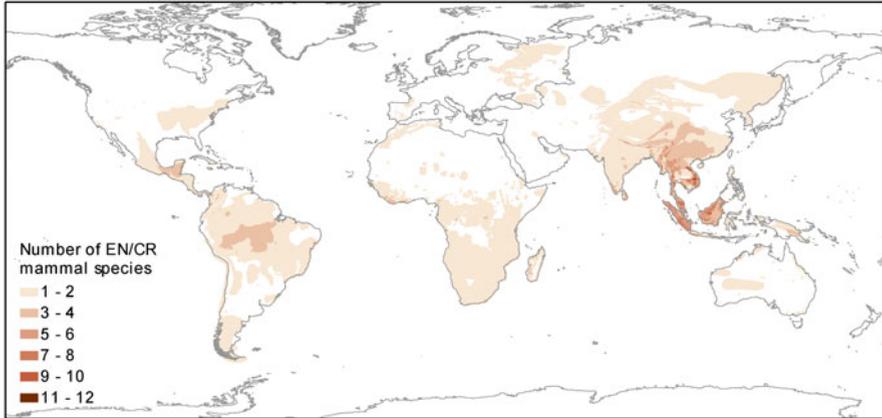


Fig. 2.1 Distribution and density of endangered or critically endangered terrestrial mammal species threatened by habitat loss and fragmentation. Distributions for several species are not visible at this scale, including two of our case study species (mountain pygmy possum and black-footed ferret). Based on data from IUCN website <http://www.iucnredlist.org>

visible on a map at a global scale (Fig. 2.1). The highest density of extant terrestrial mammal species threatened by habitat loss occurs in south-east Asia (Fig. 2.1). Specifically, the area where Laos borders Vietnam along the Annamite Chain, eastern Cambodia, central continental Malaysia and the southern coast of Sumatra have particularly high numbers of mammal species threatened by fragmentation.

The areas with the greatest amount of threatened animals as a percentage of their total mammal fauna are mostly island countries in the Indian Ocean, the south Pacific, and the Caribbean (see the IUCN analysis by Vié et al. 2009, p. 31). Bhutan, Bangladesh and India are the only non-island countries in the ‘top 20’ countries for percentage. However, the Millennium Ecosystem Assessment states that the trend is now moving away from islands with a more balanced share of extinctions being continental. Historical island extinctions were often based on species over-exploitation and the introduction of competitor species, whereas continental extinctions are more likely to be caused by habitat loss (Mace et al. 2005).

Few analyses of global patterns of mammal extinction take account of the period of human history between the advent of farming and the beginning of useful historical records of species occurrence. For example, few or no globally threatened mammals are found across most of Europe, but archaeological and historical evidence shows that wolf *Canis lupus* and lynx *Lynx lynx* were present in the British Isles (Hetherington et al. 2005; Buczacki 2002), and on mainland Europe tarpan *Equus gmelini* or tarpan-like equids (Sommer et al. 2011) and the European bison *Bison bonasus* (Benecke 2005) were all widely distributed after the most recent glaciation. IUCN data generally refers to data since 1500, but much of the land clearance for agriculture in Europe happened before then.

Taxonomic and Trait Patterns

Some mammalian orders contain a greater proportion of threatened species than others. For example, almost half of the extinct species in the IUCN terrestrial mammals dataset are from the order Rodentia. Rodents also account for the greatest number of threatened species, followed by primates. However, they are not the most at-risk order by percentage of species—63 % of the even-toed ungulates (Perissodactyla) are endangered or critically endangered, as are 60 % of the echidna/platypus grouping (Monotremata) and half of the pangolins (Pholidota) and elephants (Proboscidea). Elephants and even-toed ungulates are generally large-bodied in comparison to mammals as a whole, meaning they require a greater habitat area and many of them are exploited by humans for food. As for the largest mammalian orders, 7 % of bats and 9 % of rodents are endangered/critically endangered, as are 12 % of the Eulipotyphla (an order containing mostly small insectivores), and 10 % of the Carnivora (IUCN Red List version 2014.3).

It is complicated and difficult to assess and characterise the response of taxonomic groups to anthropogenic change because of interactions between different threats (Stork et al. 2009). Extinction risk is clumped phylogenetically, particularly for those species who are threatened by hunting rather than by habitat loss or invasive species (Fritz and Purvis 2010). However, meta-analytical studies suggest that carnivores are more sensitive to habitat area reduction than omnivores, and arboreal species are more sensitive to habitat area reduction than terrestrial ones (Prugh et al. 2008). Specialist species with narrow ecological niches are more susceptible to habitat loss and degradation, as are species with limited distributional ranges (Cardillo et al. 2005). Large bodied mammals are more vulnerable to extinction as they are disproportionately selected by human hunters (Jerozolinski and Peres 2003) and also have larger home ranges, thus in fragmented habitats are more likely to come into contact with humans (Peres 2001). Indeed, threatened mammals are an order of magnitude heavier than non-threatened ones (Sodhi et al. 2009). Generally, the generation time of a species (interval from birth to reproductive age) is related to its body mass, so larger, longer-lived, and slower-reproducing species are usually unable to compensate for high rates of harvesting and their potential for population recovery in the short term is low (Webb et al. 2002; Sodhi et al. 2009). Species which have a high intrinsic rate of population increase (e.g. high birth rates or rapid generation time) are less vulnerable, as they can compensate for greater mortality (Bodmer et al. 1997).

Present and Past Case Studies

In Table 2.1 we list examples of terrestrial mammals suffering from habitat loss, destruction and fragmentation. We find them on almost all continents and in almost every mammalian order. Other anthropogenic threats are frequently relevant

(Table 2.1). To show the variety and omnipresence of detrimental man-caused factors influencing mammals we chose several threatened species from various orders and different continents to analyse their biological characteristics and reasons for decline more thoroughly.

Asian Elephant Elephas maximus (EN)

The Asian elephant *Elephas maximus* (order Proboscidea, family Elephantidae) is the largest terrestrial Asian animal. An adult female weighs 3000 kg and an adult male weighs 4500 kg. The Asian elephant is a generalist which browses and grazes on a variety of plants and thus occurs in grassland, various types of forest such as tropical and subtropical forests, moist and dry deciduous forests and dry thorn forests as well as in cultivated areas, from sea level to 3000 m (Sukumar 2000; Choudhury 2009). An elephant's home range varies from a few tens to a few hundreds of square kilometres (Fernando et al. 2005). The Asian elephant can consume as much as 150 kg of wet-weight biomass a day (Choudhury et al. 2008). Elephants live in matriarchal family herds of 6–10 individuals; males 10–15 years old disperse and establish their own home ranges (Sukumar 2000). Their lifespan is 60–70 years (Shoshani and Eisenberg 1982). Age of sexual maturity is typically 11–14 years in females and at least 15 years in males (Vidya and Sukumar 2005). A female produces one calf every 4.5–5 years (Sukumar 2000). Average annual mortality rates over 5 years of age are 1–3 % in females and up to 6 % in males under natural conditions. However, ivory poaching can cause male mortality rates to rise up to 20 %. Annual population growth rate of the species under natural conditions does not exceed 1.5 % (Sukumar 2000).

The former range of the Asian elephant encompassed about 9 million km² while the present total area inhabited by the now isolated elephant populations covers less than half a million square kilometres (Choudhury et al. 2008). The Asian elephant population estimates of 30,000–50,000 individuals are thought to be only a crude guess (Hedges 2006). Over 50 % of the remaining Asian elephants live in India (Sukumar 2000).

Reasons for decline. The IUCN has estimated a decline of at least 50 % in the population size of the Asian elephant in the last three generations. Major threats to the species are habitat loss, fragmentation and degradation. The Asian elephant lives in the region of the world with the densest human population, which has an annual growth rate of 1–3 %, resulting in competition between humans and elephants for space. Settlement, agriculture including slash-and-burn shifting cultivation, coal mining, logging, and other activities have significantly encroached on natural habitat. Combined with land fragmentation from roads, railway lines, canals, and dams, human settlement has reduced the elephants' habitat and resulted in severe conflicts between humans and elephants (e.g. Leimgruber et al. 2003; Hedges et al. 2005; Hedges 2006). Because elephants require much larger areas of natural habitat than most other Asian terrestrial mammals, they are also among species most

vulnerable to habitat fragmentation and degradation. Moreover, their great size and large food requirements predispose them to cause large-scale crop and property damage. Over 25 years, more than 1150 people and 370 elephants were killed as a result of human–elephant conflicts in northeastern India (Choudhury 2004). People suffering from damage by elephants to their property and crops demand protection or compensation from government authorities. Lack of such reparation results in the killing of elephants in retaliation, and local antagonism toward the species and its conservation (Hill et al. 2002). In addition elephants are poached to obtain ivory, meat and other body parts (Hedges 2006). Elephants injured by poachers often retaliate by killing humans and damaging their property (Choudhury 2004) and so a vicious circle develops. By removing tusker males, poaching limits the species genetic variability and may also lead to highly skewed age and sex ratios, which can have a significant impact on population dynamics (Hedges 2006). For instance, in Periyar Tiger Reserve in southern India, the adult male:female sex ratio changed from 1:6 to 1:122 over a 20 year period (Hedges 2006).

Tiger Panthera tigris (EN)

The tiger *Panthera tigris* (order Carnivora, family Felidae) is one of the most highly ranked felids of tropical Asia in terms of the vulnerability index and, moreover, is actively threatened (Nowell and Jackson 1996). Of the nine subspecies three are already EX, two CR and the remaining four, as well as the species in general, are classified as EN.

Depending on subspecies adult females can reach up to 110–167 kg and males over 300 kg in weight (Mazák 1981). The tiger is most often found in tropical and subtropical moist deciduous forests, followed by temperate and deciduous mixed forest, tropical and subtropical dry deciduous forest, and also occasionally in coniferous forest, mangrove forest, tropical grassland and shrubland (Sanderson et al. 2006). While tigers eat a variety of animals from large ungulates or even young elephants to rodents, fish and insects, the tiger needs a healthy prey base to form a viable population (Biswas and Sankar 2002). Tigers are territorial; the size of their home range depends on the abundance of prey and varies from a tens to a few hundreds of square kilometres (Matjuschkin et al. 1980; Sunquist 1981; Barlow et al. 2011). Average age at first reproduction is 3.4 years for females and 4.8 years for males (Smith and McDougal 1991). The litter usually consists of 2–3 cubs (Sankhala 1978), and interbirth interval is about 2 years (Smith and McDougal 1991). For both males and females the mean number of offspring incorporated into the breeding population is two (Smith and McDougal 1991). In the wild the tiger may live for up to 15 years (McDougal 1991).

The historic range of the tiger extended across Asia from Turkey in the west to the eastern coast of Russia (Nowell and Jackson 1996). However, according to Sanderson et al. (2006), the tiger has lost 93 % of its range and now lives in highly fragmented populations (Fig. 2.2). The global population numbers are assessed at

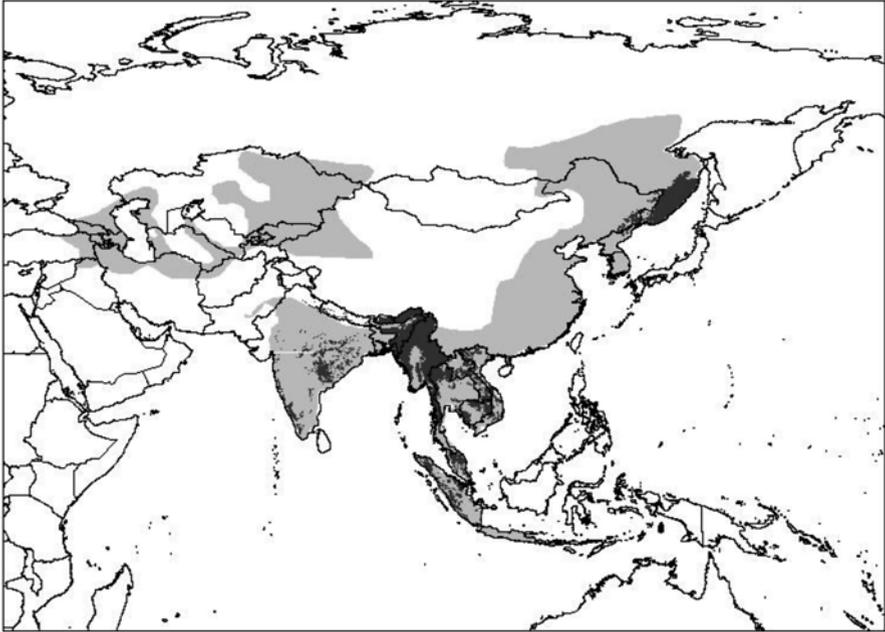


Fig. 2.2 Historical (*light grey*) and present (*dark grey*) distribution of the tiger *Panthera tigris*, redrawn from Luo et al. (2004) (with modifications) by Kate West

3200–3600 individuals, approximately half of the size estimated a decade ago (Seidensticker 2010) and the effective population size (number of reproductively successful adults) is estimated to be less than 2500 tigers (Chundawat et al. 2011).

Reasons for decline. Habitat loss, poaching of tigers for various body parts and the exotic pet trade, and overhunting of tigers' ungulate prey resulting in prey scarcity are all major threats for the tiger (Dinerstein et al. 2007; Tian et al. 2011). Like the Asian elephant, the tiger's range is a region of high density and rapid growth of human population, resulting in rapid conversion of forest habitat to agriculture and settlements combined with degradation of forests by commercial logging. Habitat fragmentation has caused isolation of tiger populations and added the threat of genetic deterioration (Wikramanayake et al. 2011).

Hunting of tigers for sport was once among the causes of population decline. Moreover, the main prey of the tiger—large ungulates—are also heavily hunted by people and have to compete with domestic animals. It is suggested that tigers may be unable to reproduce successfully at prey densities below 2–5 ungulates per km² (Nowell and Jackson 1996).

Lucrative illegal markets now exist for tiger products such as skins, teeth and claws (Nowell and Jackson 1996). Another problem is poaching of tigers for the illegal trade of their bones and other body parts used in traditional Asian medicine;

this issue includes intensive breeding, so-called farming, of tigers in China. Some authors consider this trade to be an even greater threat for the tiger than loss of habitat (Pullin 2002).

The bigger the felid, the more severe the conflict with people (Inskip and Zimmermann 2009). Tigers are thus also killed because of conflicts resulting from tiger attacks on people and livestock. In the earlier part of the twentieth century tigers were officially considered pests with bounties paid for their annihilation (Nowell and Jackson 1996). The majority of attacks on people were by unhealthy tigers, either those with physical deformities (Gurung et al. 2008) or which had been wounded (mainly by people); almost half of attacks were provoked (Goodrich et al. 2011). Removal of unhealthy tigers from the wild seems to reduce the number of human deaths (Goodrich et al. 2011), which is important as human–tiger conflicts create negative attitudes towards tiger conservation in local communities (Goodrich 2010).

***Mountain Pygmy-Possum* *Burrhamys parvus* (CR)**

The mountain pygmy-possum is a small rodent-like Australian marsupial belonging to the order Diprotodontia, family Burramyidae. The average adult weight is 40 g but varies from 30 g in spring up to 82 g in autumn, prior to hibernation (NSW National Parks and Wildlife Service 2002). The species was originally described from fossils (Broom 1896) and presumed to be extinct, but in 1966 was discovered in the Australian Alps. It is the only Australian mammal confined to alpine environments. The mountain pygmy-possum lives in patches of periglacial boulders with associated shrubby heathland (NSW National Parks and Wildlife Service 2002). The climate is harsh but the boulders provide the possum with deep, insulated hibernacula and protected nesting sites (NSW National Parks and Wildlife Service 2002). The species' diet changes seasonally and consists of invertebrates, seeds and fruit (Menkhorst et al. 2008). Survival from birth to breeding is significantly higher in females (Mansergh and Scotts 1990). Females are sedentary and live longer than dispersing males with over 11 years recorded as the maximum longevity for a wild female and only 3 years for a male (Mansergh and Scotts 1990). Habitat fragmentation is likely to decrease food availability and increase predator risk in suboptimal areas and thus skew the sex ratio (Broome 2001; Mitrovski et al. 2008). Females have only one litter of up to 4 young per year (Mansergh and Scotts 1990). They are capable of mating at 1 year of age, but only half of them will survive due to their inability to maintain fat reserves for hibernation. Juveniles spend 5 months in hibernation and adults 7 months (Geiser and Broome 1991). The mountain pygmy-possum is the only Australian mammal that depends on winter snow cover for its survival. When snow depth exceeds 80–100 cm, snow temperature at the ground level remains at the optimal 2 °C for the hibernation of the species (NSW National Parks and Wildlife Service 2002).

The former distribution of the species comprised only 2250 km², but the confirmed current range seems to cover less than 7 km² (Broome 2008). There are less than 1800 adult mountain pygmy-possums and the population is declining (Heinze et al. 2004).

Reasons for decline. The habitat of the mountain pygmy-possum is highly restricted and thus each environmental modification may have a deleterious effect. Unfortunately for the species its habitat is within the range of ski resorts and so has been destroyed and fragmented by development of roads and infrastructure for the downhill skiing industry, as well as by rock extraction for dam construction. Levelling and grooming of land for ski trails has greatly affected the amount of available habitat; in some parts as much as 80 % of the habitat has been disturbed or eliminated (Mitrovski et al. 2007). The remaining patches of good habitat are small and separated from each other by large areas of unsuitable habitat (Caughley 1986). Mitrovski et al. (2007) found limited dispersal across barriers and claim that each separate area containing the mountain pygmy-possum should be considered as an independent gene pool. In one of the Mt Butler populations, within the area of a developing resort, levels of genetic variation fell to approximately one-third of the initial level within 10 years—the most rapid loss of genetic diversity in a mammal ever recorded (Mitrovski et al. 2008). McCarthy and Broome (2000) demonstrated that mountain pygmy-possums are very sensitive to reductions in population growth rate. With a 15 % reduction in current survival and recruitment rates, the probability of decline of equilibrium populations of 20 females to 5 females or fewer within 100 years is about 90 %.

Habitat is also reduced in quality by diminishing depth and duration of snow cover. Low snow cover together with high skier and snowboarder use result in visible damage to vegetation. Winter snow grooming machinery can damage vegetation used by mountain pygmy-possum for food (Broome 2001) and may also affect hibernation of the animals by causing energetically costly arousals due to noise, vibration and changes in hibernacula temperatures and thus may decrease winter survival (NSW National Parks and Wildlife Service 2002). Another threat is predation from introduced foxes *Vulpes vulpes* and feral cats *Felis catus* (NSW National Parks and Wildlife Service 2002).

Black-Footed Ferret Mustela nigripes (in the Past EW, Presently EN) and Prairie Dogs Cynomys spp.: Joint Case Study of a Carnivore and Its Prey

The black-footed ferret *Mustela nigripes* is a North American carnivore from the family Mustelidae. It weighs 650–1400 g (Biggins and Schroeder 1988). Female black-footed ferrets reach sexual maturity at 1 year of age (Miller et al. 1996).

The species has low reproductive rates; the mean litter size at emergence of young found in the last free-living population was 3.3 (Forrest et al. 1988).

The black-footed ferret is limited to open habitat used by prairie dogs (*Cynomys* spp.) and its historical range included the grasslands and mountain basins of interior North America from southern Canada to northern Mexico overlapping the combined ranges of *Cynomys ludovicianus*, *C. leucurus* and *C. gunnisoni* (Hillman and Clark 1980). The black-footed ferret depends nearly completely on prairie dogs (Biggins and Schroeder 1988); they constitute about 90 % of its diet (Clark 1986) and the ferret also uses their burrows for shelter and litter rearing (Hillman and Clark 1980). Stromberg et al. (1983) estimated that populations of about 400–1400 *C. ludovicianus* or *C. leucurus* are needed to support the annual requirements of one reproductive female ferret and her young. However, black-footed ferrets do not considerably reduce prairie dog populations as they kill only what they can eat and prairie dogs have comparatively high breeding potential which counterbalances predation by ferrets (Hillman and Clark 1980).

Reasons for decline. This extreme dependence on prairie dogs made the black-footed ferret especially vulnerable to extinction when the numbers of its prey were greatly reduced. Conversion of prairie dog habitat to agricultural cropland was extensive and, moreover, farmers and ranchers considered prairie dogs to be pests and competitors of domestic livestock which resulted in organised state poisoning programmes to control prairie dogs. Eradication of prairie dogs began in the early 1900s or even earlier (Knowles et al. 2002). Furthermore, sylvatic plague, a non-native disease, was brought to the west coast of North America from eastern Asia in the beginning of the twentieth century, presumably on ships (Biggins et al. 2011), and became another major threat. The plague spread rapidly and caused high mortality among prairie dogs, further reducing their numbers. It was estimated that in 1960 prairie dogs occupied only 2 % of their original range. Unfortunately, ferrets also turned out to be susceptible to sylvatic plague (Miller et al. 1996). As a result of all these factors, the ferret population declined to near extinction by the late 1970s.

In 1987 the black-footed ferret was considered EW. Prior to this, 18 black-footed ferrets had been captured from the last-known population in Wyoming, in order to start captive breeding (Miller et al. 1996). Seven of those ferrets contributed unique genetic material and are considered founders of the captive population. At present there are 18 reintroduced populations but only three of them are self-sustaining (Belant et al. 2008). In 2008 there were approximately 300 captive ferrets and about 500 breeding adults in the wild (Belant et al. 2008). That genetic bottleneck means that the genetic variability of the black-footed ferrets is a major concern (Biggins and Schroeder 1988).

The future of the black-footed ferret depends on availability of prairie dogs (Jachowski and Lockhart 2009); so to conserve the ferret, prairie dogs must also be conserved at appropriate densities and distribution (Miller et al. 2007).

***Lesser Bilby* *Macrotis leucura* (EX)**

A frequent problem with extinct species is that even very basic data on them are lacking or sparse. The lesser bilby *Macrotis leucura* is an example. It was an Australian marsupial (order Peramelemorphia, family Thylacomyidae) of medium size (310–435 g, Johnson 1989). The species was confined to dry habitats in central Australia, such as dunes, sandplains with grassy hummocks, and sparse low trees and shrub (Burbidge et al. 1988). It dug out its shelters in the form of deep burrows which were important refuges from heat (Johnson 1989). The lesser bilby fed on insects, grass seeds and bulbs (Johnson 1989). Its litter consisted of one or two, rarely three, offspring (Johnson 1989).

Being previously common, the lesser bilby was drastically reduced in the early 1900s (Nowak 1991). The last specimen was collected in 1931 (Johnson 2008). According to information from Aborigines some populations possibly survived longer (Burbidge et al. 1988), maybe into the 1960s, but there is no evidence that the species still persists.

Reasons for decline. To explain numerous declines in mammals from arid lands of Australia, three main hypotheses have been proposed: introduction of exotic predators, competition from exotic herbivores and changes in fire regimes (Maxwell et al. 1996). All those factors seem to have played a part in the extinction of the lesser bilby. Predation from introduced feral cats and foxes, competition with introduced rabbits *Oryctolagus cuniculus* for burrows, trapping of bilbies for their pelt, and degradation of habitat due to changes in fire regimes are all thought to have contributed to the extinction of the species (Johnson 1989). Feral cats are supposed to have entered Australia from seventeenth-century shipwrecks on the west coast of the continent and foxes appeared later, reaching central Australia by the 1930s; native mammals seem to be unable to cope with them (Burbidge et al. 1988). The first rabbits were introduced in the 1890s and quickly became widespread (Burbidge et al. 1988).

Changes in fire regimes resulted from the depopulation of the deserts and could have caused degradation of habitat. Aborigines used fire for a variety of purposes, such as hunting and regeneration of food plants. This resulted in a mosaic of habitat types in different stages of regeneration and such environmental diversity benefited mammals. Traditional burning patterns of Aborigines also prevented the occurrence of extensive wildfires in summer. When the Aborigines left their lands, the fire regime changed to one consisting of rare but extensive summer wildfires, usually caused by lightning. This increased habitat homogeneity and restricted food and shelter availability, causing a rapid population decrease and even extinction of some species (Burbidge et al. 1988; Johnson 1989).

***Tasmanian Tiger* *Thylacinus cynocephalus* (EX)**

The Tasmanian tiger *Thylacinus cynocephalus* (order Dasyuromorphia, family Thylacinidae) is another extinct species where our knowledge contains many gaps. It was the last member of the family of carnivorous marsupials that lived in Australia.

The Tasmanian tiger was also the largest carnivorous marsupial in historic times, weighing between 15 and 35 kg (Mooney and Rounsevell 2008). Information on its prey type and size is limited (Wroe et al. 2007), but kangaroos and wallabies are suggested to have formed its main source of food (Mooney and Rounsevell 2008). The Tasmanian tiger was most often encountered in open forest and grassland, with shelters found in caves, hollow logs or dense vegetation (Mooney and Rounsevell 2008). The social structure seemed to include both stable family groups with fixed home ranges (probably territories) and nomadic solitary individuals (Paddle 2002). Breeding took place in winter or spring with two to three offspring weaned (Mooney and Rounsevell 2008). Record longevity was about 8.5 years (Collins 1973).

Reasons for decline. Several thousand years ago the Tasmanian tiger was widespread on the Australian mainland and its decline there was suggested to be caused by competition with and predation by the introduced dingo *Canis lupus dingo*. The Tasmanian tiger became extinct on the Australian mainland about 2000 years ago but survived in Tasmania (Mooney and Rounsevell 2008). However, based on anatomical details of the Tasmanian tiger, conclusions about the range of the species' prey and therefore on the possibility of competition with the dingo are ambiguous (Jones and Stoddart 1998; Wroe et al. 2007). Spontaneous predation by dingoes may be doubtful as dogs seemed to be afraid of Tasmanian tigers and almost all historical data on dogs killing Tasmanian tigers included the company of people with some kind of weapon (Paddle 2002). The Tasmanian tiger seemed to be under considerable stress long before the arrival of the dingo because of its competition with Aborigines, who also used the species as food, and the appearance of the dingo in Australia probably only sped up the process of its extinction there (Paddle 2002). There could also have been a disease introduced with alien species that affected the Tasmanian tiger (Mooney and Rounsevell 2008).

Tasmanian tigers that survived in Tasmania were probably not very numerous as Tasmania did not provide a good habitat for them (Mooney and Rounsevell 2008). After the introduction of sheep by Europeans, Tasmanian tigers were considered pests. Although the reported number of sheep killed by Tasmanian tigers was much exaggerated, extensive persecution of the species started in the first half of the nineteenth century and was reinforced by bounties (Paddle 2002). In 1936 the species was probably already extinct (Mooney and Rounsevell 2008).

European Bison *Bison bonasus* (in the Past EW, Presently VU): An Expanded Case Study and Promising Example

The European bison *Bison bonasus* (order Cetartiodactyla, family Bovidae) is the largest terrestrial mammal in Europe. Adult males weigh 440–840 kg and adult females 340–540 kg (Kraśnińska and Kraśniński 2002). The European bison is predominantly a grazer, but its diet is supplemented with some browse and bark. It prefers deciduous and mixed forests in moderate climate zones and preferentially

feeds in glades and riverside meadows (Daleszczyk et al. 2007). It is a social species: cows with calves and sub-adults form mixed groups of a dozen or so individuals while adult males live solitarily or in bull groups of 2–3 males, joining mixed groups during the rutting season (Kraśnińska and Kraśniński 2007). Home ranges of groups or solitary bulls range from a few tens to 150 km² (Kraśnińska and Kraśniński 2007). Cows start calving from 4 years of age, giving birth to one calf every second year. Bulls reach sexual maturity at 3–4 years of age, but begin to participate in reproduction 2–3 years later (Kraśnińska and Kraśniński 2007).

Extinction in the Wild, Restoration, and Current and Future Challenges for Its Conservation

The fate of the European bison is one example of how humans brought a species to the brink of extinction in a few centuries but then was able to save it through great efforts.

The European bison was historically distributed throughout western, central, and south-eastern Europe and the Caucasus. Along with the aurochs *Bos primigenius* that became extinct in 1627 it was the largest terrestrial mammal within historical times in Europe. Morphophysiologicaly the European bison is adapted to graze (Hofmann 1989) and its diet is dominated by grass and herbaceous plants (Gębczyńska et al. 1991; Kowalczyk et al. 2011). This is one aspect among others in its ecology that argue for the European bison being fundamentally an open habitat species rather than a forest species; a combination of increasing replacement of open steppe habitats by forest cover in the Holocene and increasing human pressure forced bison into remote forests for refuge (Kerley et al. 2012)—this beginning the gradual extinction process.

The bison was hunted as a considerable food resource but also because it competed with domestic stock and arable farming. Overharvesting and increasing habitat loss since settlement of the human population led to a continuous reduction of the range of the European bison. By the early Middle Ages the range of the bison was already dramatically fragmented to small relict populations.

The European bison was one of the first species to be protected by law, mainly to serve the rulers of the area as a challenging and majestic game species. The bison as charismatic game was so popular that early attempts were undertaken to reintroduce bison to habitats where it had already become extinct, for example, in Mecklenburg in 1689 and in Saxony in 1733 (Pucek et al. 2004). These early reintroduction projects failed due to poaching (Tillmann 2008).

In the early twentieth century only two populations survived in the wild: in Białowieża Forest (over 700 individuals of *B. b. bonasus*, Wróblewski 1927) and in the western Caucasus mountains (between 400 and 500 *B. bn caucasicus*, Heptner et al. 1966). These remaining two populations were finally exterminated within a very short time in 1919 and 1927, respectively, as a result of warfare and extensive poaching. Only 54 individuals survived in a few European zoological gardens.

This precarious situation of the European bison and the urgent need to rescue it from extirpation was reported to a broader audience in 1923 by the Polish zoologist Jan Sztolcman at the first International Conservation Congress in Paris. In consequence, 16 participating nations founded the International Society for the Conservation of the European Bison in order to coordinate a breeding programme to increase the population size and to maintain the remaining genetic variability, with the aim of re-establishing free-ranging herds in the European bison's former range. The most important tool in achieving this goal has been the European Bison Pedigree Book (EBPB) which was first published in 1932 and continues today. It is the first documentation of the breeding stock of an endangered species used as a basis for its conservation programme. The greatest challenge of the EBPB is the conservation of the remaining genetic variability as the bison went through a severe genetic bottleneck—the entire contemporary genetic variability derives from only 12 founders—and inbreeding remains as a major threat to the world population (Tokarska et al. 2011).

The breeding programme started in 1929 in Białowieża with a slow but steady increase in the captive world population, succeeding such that in 1952 the first reintroduction to the wild was launched in Białowieża Forest. Since then, additional captive and free-ranging populations have been founded across Europe. By 2014 the global population was a little over 5200 bison, of which 69 % were free ranging but distributed across 35 isolated and generally small populations in Poland, Belarus, Lithuania, Russia, Ukraine, Slovakia, Romania and Germany (Raczyński 2014). Although the population of the European bison has grown during the twentieth century, the species still faces an uncertain future. Consequently in Europe, the bison is included in Appendix III (protected fauna species) of the Bern Convention on the conservation of European wildlife and natural habitats and is classified as an endangered species (VU: D1) by the 2014 IUCN Red List of Threatened Species. The European Union recognised this special responsibility and listed the European bison as a strictly protected priority species of community interest, whose conservation requires the designation of special areas of conservation (EU Habitats Directive, annex II and IV).

With its current status the world population of European bison is still not regarded as saved from extinction (Pucek et al. 2004). A great danger is the further loss of genetic variability. Most free-ranging bison populations have less than 50 individuals, with only ten of the 35 free-ranging populations numbering over 100 individuals (Raczyński 2014), additionally the populations are geographically isolated. For this reason most free-ranging populations are prone to catastrophic events such as epidemics and extinction and thus their protection should include the establishment of metapopulations and interventions to provide gene exchange (e.g. Daleszczyk and Bunevich 2009; Kuemmerle et al. 2011).

The range of the free-ranging populations is usually limited by habitat suitability but much more by acceptance (Lawreszuk 2012). The know-how and the will to coexist with European bison and large mammals in general have been lost over the centuries. In order to integrate this species into today's anthropogenic landscapes, conservation faces manifold challenges.

Two recent projects—one in Poland and one in Germany—are pilot schemes to develop new sustainable conservation programmes for free-ranging bison populations.

The EU-Life project 'BISON LAND—European bison conservation in the Białowieża Forest, Poland' is a sensitive approach integrating local to regional land-use interests, to improve habitat suitability and habitat size for the biggest free-ranging European bison population. This should form a basis for further dispersal and a sustainable population increase as part of a wider ecological network (Kowalczyk et al. 2010).

The BISON LAND project has implemented a set of public awareness-raising activities and published attractive dissemination materials. Various environmental activities were accompanied by public consultations and information-education campaigns that enhanced knowledge and awareness of the region's population of this species. The bison was promoted as a tourist attraction in the region (Lawreszuk 2012). Conflict management, e.g. the protection of agricultural areas affected by bison or managing damage-causing individuals, was an important tool in increasing acceptance by the local population. Building or equipping farmers with fences and catching troublesome individuals to transport them deep into the forest were the most effective measures (Kowalczyk et al. 2010). Compensating damages to crops and meadows but moreover contracting meadows from local farmers for the needs of bison were also found to be very effective tools for improving tolerance, as this actively involved farmers in bison management as part of their income and their local environment.

Beside the optimisation of the human dimension in bison management, various other actions have been implemented focusing on the improvement of its living conditions. For example, feeding sites, forest meadows and watering places have been created as well as existing forest meadows managed specifically as ideal grazing sites for bison. These practical management tools were implemented in a spatially strategic way in order to guide bison to ecological corridors connecting their current home range with further suitable habitat.

As a result of the comprehensive conservation measures, the bison population increased in size and range, damages and conflicts with land-use were reduced and empathy for free-ranging bison among the local community was raised. The concept of extending the bison range by the designation of ecological corridors could, on this basis, then be implemented in the Provincial Land Development Plan. This allowed free migrations so that gene flow between bison groups became possible. Many problems of the 'refugee species', as the European bison was described by Kerley et al. (2012), could be mitigated by the management model applied in BISON LAND.

In today's reintroduction attempts of European bison the focus lies not only on the conservation of this endangered species per se but also on the reintroduction of its ecological role and its interactions with the environment that have gone extinct alongside local extinction of the bison. However, until now, there had never been a serious attempt to reintroduce a population of free-ranging bison to its former range in western Europe. The initiative to reintroduce the European bison in an intensively

managed commercial forest in North-Rhine Westphalia, Germany, was formulated by various stakeholders in 2003 and finally happened in 2013 (Tillmann et al. 2013). The reintroduction of a small population of European bison is at this stage unique for western Europe and aims on the one hand to contribute to the conservation of this highly endangered species and on the other hand to fill again its abandoned ecological niche in a central European forest landscape. This innovative reintroduction project relies on bison management experiences gathered in the free-ranging populations in Eastern Europe, particularly the modern management approaches as developed within BISON LAND. Beyond that, this attempt faces challenges that are different or additional compared to those faced by the eastern European populations.

One very important pillar of this project was to inform and involve the public and relevant stakeholders from the very beginning. In this context, the initial consultations and information events for local to regional stakeholders, authorities and the wider public revealed several potential points of conflict, particularly arising in the fields of agriculture, forestry and tourism. To counter these concerns, a feasibility study was conducted to evaluate habitat suitability and habitat capacity for the European bison. Experts on various free-ranging bison populations in eastern Europe were consulted in order to incorporate existing knowledge and the project area was found to be suitable for carrying a free-ranging herd of European bison (Lindner et al. 2010; Tillmann et al. 2012). After these comprehensive information campaigns, questionnaire surveys and interviews among the local community revealed a broad acceptance of the idea to reintroduce European bison in this region (Decker et al. 2010).

The reintroduction itself began in 2009 with a thorough Environmental Impact Assessment and programmes of stakeholder participation and conflict management. During the initial captivity phase, European bison were found to be manageable in the Rothaargebirge mountain range and therefore permission to release the herd was given by the state of North Rhine-Westphalia in the winter of 2012/2013. This was a milestone in efforts to conserve this species and their ecological role in human-dominated landscapes and can be taken as an exemplar for reintroduction projects elsewhere.

Both the BISON LAND and the Rothaargebirge project have shown that bison not only shape their natural environment but moreover their presence in the wild has an outstanding human dimension. The paradigm of 'conservation with development' has attracted increasing support from conservationists as conservation efforts often lack money and the economic value of wildlife is being taken into account more and more. The integration of human development needs with bison conservation objectives can result in the establishment of mutually beneficial relationships as can be observed in almost all free-ranging bison populations. Bison based ecotourism ventures can earn direct revenues for local and regional communities. Besides the economic value of this charismatic species it has an additional educational value.

Summing up, the European bison plays not only a significant role as a flagship species in the ecosystem but moreover has an aesthetic, cultural and economic value

that should be considered and integrated into regional development programmes and marketing concepts. The conservation needs of the European bison can be described as representative for many other large mammals, in the same manner as the need for the acceptance and support of local people can. As demonstrated in the two flagship projects in Poland and Germany the human dimension plays the strongest role in establishing a sustainable coexistence of people and bison in anthropogenic landscapes.

For the long-term conservation of the global population of European bison it is essential to increase the population further. Its broad ecological valence would allow the opportunistic European bison to inhabit large areas within its former area. However, against the background of conflict avoidance and acceptance, particularly in the context of agriculture, forestry or traffic, suitable habitats are reduced dramatically and few areas remain for potential reintroductions. Reintroduction attempts need to be well prepared in order to be accepted by the local community. Pilot projects as described above are of outstanding importance in developing sound and adaptive management concepts to serve as models and 'icebreakers' in facilitating further reintroduction initiatives elsewhere. Additionally these projects assist in learning anew the routine coexistence with a large herbivore in the same ecosystem.

Conclusions: What Can We Learn?

The history of various species shows that it is quite easy to decimate a species and very difficult to get it back—a truism but so often forgotten. Our analyses, case studies, and the examples of threatened or extinct mammals given in Table 2.1 all indicate that there are usually several detrimental factors acting in parallel or synergy, which contribute jointly to the decline or extermination of the species. Because of habitat loss and degradation, many species of both herbivores and carnivores have to compete with humans for space and food. The competition of larger species with people may take the form of direct conflicts, as in the case of the Asian elephant or the tiger, which have lost most of their habitat and must therefore obtain food within a limited area shared with a dense human population. Some species are considered a threat to crops or domestic livestock and the magnitude of their persecution, often disproportional to the actual damage they cause, may bring the species to the brink of extinction (Tasmanian tiger, prairie dogs) and may also influence ecosystem equilibrium and thus indirectly harm other species dependent on those being persecuted (black-footed ferret). Extinctions of species may disrupt essential ecological processes such as seed dispersal and thus lead to cascading losses, ecosystem instability, and a higher general rate of extinction (Sodhi et al. 2009). The loss of entire functional groups, for example, frugivores, is likely to have severe consequences for ecosystem function (e.g. Grelle 2005). Trophic cascade effects have also been demonstrated in a wide variety of systems, where removal or reintroduction of a single apex predator species causes a trickle-down effect throughout the

entire ecosystem (Ripple et al. 2014). Introduction of alien species or diseases caused significant problems for the survival of half of the species described in our case studies (mountain pygmy-possum, black-footed ferret, lesser bilby, Tasmanian tiger). Political instability, both war and civil unrest, frequently resulting in extensive poaching, forms another major threat which is often underestimated (European bison). All the species from the case studies have low reproductive rates so their potential for recovery is low, and those which are also habitat specialists (black-footed ferret) or species with highly restricted ranges (mountain pygmy-possum) have even less chance of surviving (Sodhi et al. 2009). A species that was close to extinction usually has a severely limited gene pool and so it may never be completely safe in the future (the European bison, black-footed ferret).

If we want to preserve biodiversity in our world, a compromise has to be reached between the needs of people and the needs of nature. Global biodiversity has exceptional value, for which substitutes cannot be found. Biodiversity gives us opportunities for education, relaxation and pleasure, and also for practical use. Sustainable use of biodiversity as a renewable natural resource may help in its conservation, as this means that species are not over-used and both people and the environment benefit from it. Individual conservation programmes are necessary for each threatened species based on a clear understanding of species' needs and threats. Protection of a species has to also include protection of its feeding base. Coexistence of humans and large animals within the same area may be sometimes difficult but is possible. Among solutions proposed to reduce dramatic conflicts with problematic large species are:

- Maintaining larger intact habitats at the cost of smaller fragmented areas.
- Development of agricultural lands outside of areas where they may attract large herbivores (for example, not on the way to a watering place).
- Creation or enlargement of protected areas for conflict-arousing species to separate them from people while providing them with food and space.
- Securing ecological corridors among the areas to create metapopulations and allow gene exchange.

In places where conflicts are rather seldom, a system of compensating for damages done by the species may be a good solution. In our times of fast development, human acceptance of the close proximity of sometimes problematic wildlife is often as important as suitable habitat; therefore, education, especially for local communities, is vital to change their negative attitude and convince them of benefits connected with the species' presence. An interesting solution, used in many African countries, is a system of returning benefits from wildlife resources (such as tourist park entrance revenue) to rural communities within community-based natural resource management schemes (Zhang and Wang 2003). This is intended to motivate people in rural areas to discourage poaching inside protected areas and to protect wildlife resources outside them. It is also recommended to involve local communities in programmes to keep the problematic species away from their crops or homes, such as maintaining barriers (MacKenzie 2012), as well as in the management of the given population, for example, in capture and translocation of aggressive individuals

(Sukumar 2000). It is essential to involve local people in protection and management of threatened species because nobody will take better care of them and nobody could be a worse enemy for them than the people living in the same area.

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