CHAPTER 6

People and wild felids: conservation of cats and management of conflicts

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A lioness killed in an illegal wire snare by poachers. ©. A.J. Loveridge.

Introduction

Wild felids and people have a complex and often paradoxical relationship. On the one hand, humankind admires and reveres felids; cats appear as cultural icons and symbols across the ages. In addition, there is a growing awareness of the value of wild felids as key components of ecosystems, tourist attractions generating income, umbrella species for conserving ecosystems, and flagships for engendering public support for conservation. These positive values sometimes contrast strongly with the relationship between wild felids and people in areas where they coexist. Human conflicts with wild cats, overexploitation of felid and prey populations, and habitat loss and fragmentation have extirpated felid populations and still threaten many more. The ways in which people value and interact with organisms and their habitats is at the heart of conservation. This chapter explores some of the interrelationships between people and wild felids, where human actions threaten felid populations.

Why conserve wild felids?

We preserve carnivores for aesthetic, symbolic, spiritual, ethical, utilitarian, and ecological reasons. Felids are culturally valued and are important as cultural icons and symbols. Felids are widely depicted in art, from Stone Age petroglyphs and cave paintings to more modern depictions of cats as art, reminding us that humans have interacted with felids for as long as we have been humans. Human culture is enriched by the symbolic use of felid images. Felids adorn many currencies, are heraldic symbols on coats of arms and appear widely on the badges of national sports teams. They were revered by many cultures, for instance, the lion-headed goddess Sekhmet represented war and disease in ancient Egypt. The jaguar (*Panthera onca*) features significantly in Central American cultures. Similarly, the tiger (*P. tigris*) is culturally important in South and East Asian cultures (Weber and Rabinowitz 1996).

Perhaps because of their place in our consciousness and the ease with which they are recognized, felids are used as umbrella and flagship species to conserve habitat, benefiting both felids and biodiversity generally. Large carnivores are often highly mobile and viable populations require large tracts of suitable habitat and adequate prey populations. Thus, strategies for the protection of large felids also offer protection for large 'functioning ecosystems' (Soulé and Simberloff 1986; Noss *et al.* 1996; Seidensticker *et al.* 1999). It is much easier to motivate the public and governments to protect a charismatic carnivore than a species that is seldom seen and outwardly unremarkable. Examples of this include the protection, through an initiative called *Paseo Panthera* (the Path of the Panther), of biological corridors on the Panamanian Isthmus, used by jaguars, pumas (*Puma concolor*), and other wildlife. 'Tiger Reserves' protect significant areas of biodiversity in India, as well as halting the decline of tigers, and tigers are used as a flagship to promote conservation efforts along border regions in the Indian subcontinent and Indochina (Weber and Rabinowitz 1996; Rabinowitz 1999; Seidensticker *et al.*, Chapter 12, this volume; Sunquist and Sunquist 2002).

Moral philosophers and animal welfarists are united in their arguments that every species has an intrinsic conservation value that implies it has a right to survive. Intrinsic conservation values have served as an impetus for conservation. However, intrinsic value in itself has been insufficient to secure successful conservation because humans are motivated more by economic self-interest than by ideas (Kellert *et al.* 1996). Economic benefits can provide powerful incentives to local communities to protect biodiversity. Felids are economic assets and, when used sustainably through tourism, trophy hunting, or commercial exploitation can contribute substantially to both their own conservation and that of their habitats. For example, African lions (*Panthera leo*) earned Amboseli National Park, Kenya, US $27,000 per lion per year in tourism revenues (Western and Henry 1979). Trapping of furbearers or unregulated trophy hunting can lead to controversial declines in populations. However, where it is carried out on a well-managed and sustainable basis, it can provide incentives for long-term protection, a good example being sustainable harvests of Canada lynx (*Lynx canadensis*) furs. Finally, felids play an important and often regulatory role within the ecosystems they inhabit (Mills and Biggs 1993; Karanth and Stith 1999; Grange and Duncan 2006). This ecological role needs to be factored into any valuation we make of felids in the wild. It must also be included in any assessment we make in valuing our relationship with nature and the pristine habitats that should form the baseline for all conservation efforts.

How people impact felids: anthropogenic threats to felid populations

Habitat loss is a global phenomenon, affecting all species. The earth's human population has increased from 3 billion to 6 billion since 1960. The global economy has increased sixfold and food production increased by 2.5 times. Nearly 25% of the earth's terrestrial surface is now under cultivation. The past 40–50 years have seen a sharp increase in the amount of land converted to agriculture, and projections suggest that further conversion is to be expected in the future (Millennium Ecosystem Assessment 2005). Conversion of natural habitat to agricultural land, urban development, and destruction or fragmentation of habitats through logging, building infrastructure (e.g. dams, roads, power lines, oil, and mineral extraction), and other human activity has serious impacts on wild felid populations. Equally important is loss of prey populations through over-hunting, retaliatory persecution, habitat loss, or fragmentation. Predator population densities are closely correlated with prey-population biomass, thus loss of prey species (either through overexploitation, such as the bushmeat trade, or habitat destruction) causes linked declines in felid populations (Karanth *et al.* 2004c; Henschel 2007).

Against a backdrop of global habitat loss, felids face a number of proximate anthropogenic threats. One of the most important ways in which people impact felids is through increasing rates of non-natural mortality (Fig. 6.1). For felid populations below their carrying capacity, anthropogenic mortality is thought to be largely additive rather than compensatory to natural levels of mortality (Lindzey *et al.* 1992). By contrast, in highly protected populations anthropogenic mortality is rare, for instance only one (7%) of 14 leopard (*Panthera pardus*) deaths recorded in Kruger National Park, South Africa, by Bailey (1993) was due to poaching; the rest were due to intraspecific fights, predation, and starvation. In general, anthropogenic mortality in many studied felid populations is high. While this may in part be due to the fact that conservation biologists choose to study populations that are under threat, it is also an indicator of the impact that humans have on felid populations. Common sources of anthropogenic mortality are legal hunting and trapping, poaching, problem animal control (both legal and illegal), and vehicle accidents. Furthermore, felid populations can also be exposed to diseases carried by domestic carnivores, often as a result of human encroachment into wild habitats. In some populations, anthropogenic mortality can be extremely high and lead to population declines. For instance in Laikipia, Kenya, 17 of 18 tagged lions which died were killed in retribution for livestock raiding, with the population declining by about 4% per annum (Woodroffe and Frank 2005). Similarly, lion density was significantly reduced by conflict with local people on group ranches surrounding Masai Mara Reserve, Kenya (Ogutu *et al.* 2005). In a study of Amur Tigers, all seven recorded deaths of individually recognized adult females were due to poaching and 57% of cub mortality in this population was anthropogenic (Kerley *et al.* 2002, 2003).

Utilization of felid populations varies substantially in intensity, with mortalities from this source ranging from around 7% in Eurasian lynx (*Lynx lynx*) in Switzerland (Schmidt-Posthaus *et al.* 2002a) to 27 (62.7%) of 43 deaths of radio-collared lions around Hwange National Park, Zimbabwe (Loveridge *et al.*, Chapter 11, this volume). Trophy hunting made up between 11.7% and 50% of recorded mortality in pumas (Logan *et al.* 1984; Lindzey *et al.* 1988; Cunningham *et al.* 2001), 37.5% in radio-tagged leopards in Natal, South Africa (Balme and Hunter 2004), and trapping

Figure 6.1 Lioness killed by a steel wire snare. Wire snares set for both predators and prey species may increase levels of mortality within predator populations. (Photograph courtesy of P. Lindsey and Sango Ranch, Save Valley Conservancy.)

made up around 45–66% of mortality of marked bobcats (*L. rufus*) in Mississippi and Maine, United States (Litvaitis *et al.* 1987; Chamberlain *et al.* 1999). Trapping mortalities made up 19 (70%) of 27 recorded deaths of Canada lynx (*L. canadensis*) in the Northwest Territories (Poole 1994). However, for many exploited populations few demographic data are available and this is particularly true of some populations hunted and trapped for the fur trade (e.g. spotted cats in South America in the 1960s and 1970s).

Conflict with people can lead to high levels of mortality, particularly where predator eradication occurs. For instance, an average of 29 (range 10–42) lions are destroyed each year in farmland around Etosha National Park, Nambia, where they pose a threat to livestock, with 563 lions killed over a 20-year period (Stander 2005b). In puma populations monitored in Arizona and Utah, United States, 47% and 24%, respectively of marked animals that died were killed in defence of livestock (Lindzey *et al.* 1988; Cunningham *et al.* 2001). Around 3500 pumas per year are killed by people (through sport-hunting, protection of livestock, and vehicle accidents) in the western United States (Papouchis 2004).

Poaching can contribute substantially to mortality within a population. Ferreras *et al.* (1992) found that 42% of deaths of Iberian lynx (*Lynx pardinus*) in Donana National Park, Spain, were due to illegal poaching. Poaching accounts for a significant proportion of tiger mortality in India and Russia (Kumar and Wright 1999; Miquelle *et al.* 2005a) and Kenney *et al.* (1994, 1995) suggest that even moderate levels of poaching over relatively short time periods (~6 years) can lead to massive population declines of up to 95% in this species. In the Russian Far East, Chapron *et al.* (2008b) suggest that tiger populations cannot recover if annual mortality rates exceed 15%. Finally, road accidents can also be significant sources of mortality in some felid populations. Forty-five per cent of ocelot (*Leopardus pardalis*) mortality in Texas (Haines *et al.* 2005) and 48% of recorded puma deaths in Florida were due to road accidents (Taylor *et al.* 2002).

In this chapter we identify two key areas where felid populations face anthropogenic threats. Firstly, we explore the way felids impact people's lives and livelihoods and in turn the way in which people retaliate. We then consider the impacts of trade and overexploitation on the conservation status of felid populations. In each section, we offer a synthesis of the potential conservation and management solutions.

Conflicts between felids and people

Key areas of conflict between people and felids are through depredation on domesticated animals or game species and, less frequently, when large felids kill or injure people. Livestock raiding or human deaths often lead to retaliatory killing of the felids responsible. We will discuss, in the sections below, the issues of livestock depredation and human killing, the consequences for felid populations, and potential management solutions in situations where such depredations occur.

Consequences of conflict for felid populations

Loss of human life or livelihood provides the impetus for people to attempt localized or sometimes wide-scale eradication of predators. Historically, eradication of carnivores has been a state-supported priority, incentivized by rewards and bounties. From 1860 to 1875, 4708 tigers and leopards were reported killed in India (Boomgaard 2001). In the United States, over 1160 pumas were culled from 1987 to 1990 in an attempt to limit livestock losses in ranching areas (Johnson *et al.* 2001b). Wide-scale lethal control of predators still occurs in some frontier areas. For example, Michalski *et al.* (2006a) found that 110–150 jaguars and pumas were killed over a 12-month period by professional predator hunters in southern Amazonia, Brazil.

Contemporary government policies regarding predator management are generally more enlightened. However, in areas where depredations are perceived to impact livelihoods, both legal and illegal retaliatory killing of felids can extirpate populations. Frank *et al.* (2006) describe the decimation of the lion population over large areas of Kenyan Masailand, where lion populations had previously been secure. The increasingly negative attitudes to wild predators in Masailand are at least nominally due to livestock depredation. However, limited involvement by local people in wildlife tourism and lack of access to the revenue generated, coupled with ready access to agricultural poisons, may also be motivations behind increasing levels of illegal predator control. In Kenya, today, the use of agricultural insecticide Furadan (carbofuran) to kill predators is probably the single greatest source of lion mortality; in Laikipia and Kajiado districts alone, a minimum of 70 lions are known to have been poisoned since 2001 (L.G. Frank, S. Maclennan, and A. Cotterill, unpublished data) and populations of vultures and other scavenging birds are plummeting throughout the country (S. Thomsett, personal communication).

Predation by felids on ungulate game species

Large felids can come into conflict with people over competition for wild prey species. Predators were even persecuted as vermin in some national parks until as late as the 1950s, because carnivores were thought to suppress 'game' numbers (Davison 1967; Smuts 1976). A worldwide meta-analysis of human–predator conflict found that predators are reported to kill from 0.02% to 2.6% of livestock, but 9% of game species annually (Graham *et al.* 2005). Where wild ungulates are utilized by people for either commercial gain (commercial hunting and game farming) or for leisure (sport hunting), competition and conflict may occur between users and large felids. For example, cheetahs in Namibia kill economically valuable wild ungulates on commercial game farms, leading to persecution of the cats by farm owners (Marker *et al.*, Chapter 15, this volume). Similarly, hunters in the Russian Far East come into conflict with tigers over real or perceived competition for ungulates (Miquelle *et al.* 1999a). European lynx are persecuted when they compete with sport hunters for prized ungulate trophy species (Breitenmoser *et al.*, Chapter 23, this volume).

Livestock depredation by felids

Felids are obligate carnivores. Humans and felids come into conflict in ecosystems, where a high proportion of ungulate biomass is made up of domesticated species. For the most part, the smaller felid species do not cause economically significant losses to domestic livestock, although species such as serval (*Leptailurus serval*), wildcat (*Felis silvestris*), and guiña (*Leopardus guigna*) may occasionally prey on small stock or poultry (Sanderson *et al.* 2002c; Sunquist and Sunquist 2002). Generally, species ranging in size from the caracal (*Caracal caracal*; ~12 kg) to the size of the tiger (~235 kg) have been found to be the most problematic stock raiders. With the exception of the caracal and Eurasian lynx (~23 kg) all of the most import livestock-killing felids are 50 kg or heavier (Inskip and Zimmermann 2009). Table 6.1 provides some examples of livestock depredation by felids. Comparison of trends between studies is complicated by geographic and ecological differences, varying spatial and temporal scales of the studies, and reporting of widely different parameters and data. This makes rigorous evaluation of trends difficult; however, there are a number of broad patterns that emerge and these are discussed in more detail in the following text.

Patterns of livestock depredation

Different size classes of livestock are vulnerable to different suites of felid predators. Smaller felids, such as lynx and caracals, prey on the smaller size classes of livestock such as sheep and goats, while the largest felids prey on the full size range of livestock. Intermediate-sized felids, such as pumas and leopards prey on smaller livestock and juveniles of the larger species.

The spatial distribution of livestock depredation is often uneven. Stahl *et al.* (2001a, b) found that in the French Jura Mountains, there were consistent 'hot spots' of lynx predation on sheep. The areas were often also areas with high roe deer (*Capreolus capreolus*) densities, suggesting that lynx populations may have been sustained by reservoirs of natural prey. Similarly, in southern and central Brazil, livestock were most vulnerable to puma and jaguar predation within or in close proximity to forest fragments or riverine forest, habitats that provide felids with cover and residual levels of natural prey (Mazzoli *et al.* 2002; Palmeira *et al.* 2008). In southern Africa, livestock predation by lions and leopards is often higher close to the boundaries of protected areas (Fig. 6.2). This was found to be the case adjacent to Khutse Game Reserve, Botswana (Schiess-Meier *et al.* 2007) and Chirisa Safari Area, Zimbabwe (Butler 2000). In

Table 6.1 Examples of studies of depredation by large felids on livestock, illustrating the nature, extent, and economic impact of the problem across four continents. (See notes at end of table for livestock species codes.)

Location	Dates	Predator species	Livestock species	Livestock damage	% total livestock depredated	Other causes of livestock loss	Economic impact	Details
Botswana, Kweneng district[a] (38, 123 km^2)	2000–02	Lion, leopard, and cheetah	C, G	857 leopard attacks, 588 lion attacks, and 211 cheetah attacks.	0.34% of all livestock in area (1.0–11% depending on farm size)	2.8–12.6% lost to disease, starvation, and accident		Losses mostly of free-ranging livestock. Depredation 10 times higher close to reserve. Lions and leopards caused 64% damage
Kenya, ranches adjacent to Tsavo East NP[b] (690 km^2)	1996–99	Lion and cheetah	G, S, C and Do, Ca	433 head in 312 attacks, lion responsible for 85.9%, and cheetah 2.9%	2.4% of stock holdings		Value of total loss (2.6% of herd's economic value): US $8749/year; Cost per lion: US$290 p.a.	Seasonal peak in depredation corresponded to low-prey density during wet season. Problem lions shot, of which 66% were subadult males
Kenya, Laikipia[c] (1443 km^2)	1999–2000	Lion, leopard, and cheetah	C, G, S	Lions killed 143 cattle, 396 shoats; leopards killed 38 cattle, 58 shoats; cheetah killed 94 shoats	0.7–0.8% cattle and 1.4–2.1% sheep	2.5% cattle, 8.2% sheep/ goats to disease and stock theft	US$360 to maintain a lion, US$211 for a leopard. US$40/ household lost on group ranches (11% of annual income)	Lethal control by commercial ranchers of stock raiders was focused on culprits. Subsistence pastoralists less tolerant. Lion population declining by 4% p.a.
Kenya, Group Ranches, adjacent to Masai Mara GR[d] (1500 km^2)	2003–04	Lion, leopard	C, G, S	130 attacks, 147 stock animals killed Lions responsible for 15% of attacks, leopards 32%	Overall loss of 0.2% cattle and 0.6% shoats. Of cattle losses, 57% due to lions		Loss of US$1866/year to lion, US$984 to leopard	Attacks more frequent in wet season, and correlated with lower prey density. Dogs effective at deterring attacks

Kenya, Loldaiga Ranch, Laikipia[e] (2000 km^2)	1989–95	Leopard, Lion, and cheetah	C, S	Each year lions kill on average 15.8 cattle, 6.3 sheep; leopards 4.3 cattle, 10.5 sheep; cheetah kill 9.8 sheep	Total loss to felids: 0.45% cattle and 0.67% sheep	Losses to theft larger than predation losses	Lions cost US $14,400/year (US $0.72/ha/year); leopards US$5000/ year (US$200/leopard, US$0.25/ha/year)	Losses and costs of tolerating predators were low. High wildlife density may buffer against livestock killing
Kenya, Kitengela Conservation Unit[f]	1974–75	Lion	C, G, S	11 cattle and 14 shoats killed 1974–75	Total loss to felids: 0.05% cattle and 0.07% shoats			1970–75, lions killed 92 head. 19 lions shot, 2 speared in retaliation for stock killing
Cameroon, villages adjacent to Waza NP[g] (1700 km^2)	2002	Lion	C, G	118 (±0.75) head/year		Loss to disease similar to depredation	US$37–1115 (mean US$589)/owner/year	Predation was seasonal. Higher close to NP boundary, where predation occurred all year round
North-east Namibia, Tsumkwe District[h] (4869 km^2)	1992–95	Lion and leopard	C, H, D, P	Lions killed 20 cattle and 5 horses; leopards killed 43 cattle (calves), 32 dogs, and 11 chickens	4.3% cattle, 12.2% horses lost to lions; 9.2% cattle, 23% dogs, and 2.4% poultry lost to leopards		N$83/village to lions; N $55/village to leopards	95% attacks at night when stock left in bush. 14 lions translocated, 2 shot; 6 leopards translocated
Zimbabwe, Gokwe communal land[i] (33 km^2)	1993–96	Lion and leopard	G, C, Do, S, Pi	Lions killed 81 head; leopards killed 30 head (mostly goats and cattle)	Predation by all predators reached 5% of livestock	5.8% lost to disease and starvation	US$13/household/ year (12% annual income); lions cause US$2640, leopards US $580 worth of damage/ year	Attacks mostly in dry season. Lions responsible for 34% predation, 57.5% of economic loss, and killed 3.7 animals per attack (most from Kraals)

(*Continued*)

Table 6.1 (Continued)

Location	Dates	Predator species	Livestock species	Livestock damage	% total livestock depredated	Other causes of livestock loss	Economic impact	Details
Bhutan, Jigme Singye Wangchuck National Park[j]	2000	Tiger and leopard	C, Y, H, S, Pi	76 domestic animals (of which 63 cattle) lost to predators	2.3% of livestock lost (to all predators including dhole and black bear)		Overall: US$44.72/person/year (17% per capita income in area). For households affected: 84.5% of cash income	Tiger and leopard cause 82% of economic loss. Predation appeared higher close to forests and when stock poorly protected
India, Gir Wildlife Sanctuary and surrounds[k]	1986–91	Lion	C, B	Average of 1265 head/year inside GWS, 1691 head/year outside	0.4% of stock in 'tehsils' adjoining Gir	High levels of starvation during drought		Very high densities of livestock kept in areas around Gir. People are relatively tolerant of losses
India, Indian Trans-Himalaya[l] (30 km^2 and 12,000 km^2)	1995–96 and 2002–03	Snow leopard	Y, C, G, S	1.6 head/household 1995–6; 0.6–1.1 head/household 2002/3	12% of livestock holdings/year in region (1995–6)		US$128/family (50% average per capita income)	High losses in areas with low natural prey. Livestock accounts for 40–58% of snow leopard diet
India, Bhadra Tiger Reserve[m] (495 km^2)	1997–99	Tiger and leopard	Mostly C	219 head killed in 131 incidents	12.9% of livestock in household	3.1% lost to starvation and disease	16% of annual income/household	Losses lower in cropping season when people out in fields
Nepal, Annapurna Conservation Area[n]	1988–89	Snow leopard	G, S, C, Y	0.6–0.7 animals/household/year (mostly goats)	2.6% of total stock		25% (US$42) of annual per capita income	People highly intolerant. >7 snow leopards known to have been killed in 10 years

France, Jura Mountains[o] (6000 km^2)	1984–98	European lynx	S	1782 livestock killed (mostly sheep); 92–194 sheep/year (1990–98)	0.14–0.59% of sheep at regional level			High predation in 0.3–4.5% total area. Attacks increased as lynx population increased (3 attacks in 1984, 188 in 1989). 10 problem lynx removed
Southern Brazil, Santa Caterina[p] (138 km^2)	1993–95	Puma	C, S, G, Pi	Pumas killed 9 cattle, 172 sheep and goats and 5 pigs	0.1% of cattle holdings, 23.8% of shoats, and 3% pigs	2.2% cattle, 2.6% shoats, 3.8% swine lost to disease, theft, and accidents	Cattle US$1890 (0.27% total value), sheep US$5900 (32% value), goats US$4332 (38% value)	Managed flocks had lower losses than free-ranging flocks. >27 pumas killed 1988–95, often illegally
Brazil, Alto Florista, Mato Grosso[q] (34,200 km^2)	2001–04	Jaguar and puma	C	72% of predation by jaguars, rest by pumas	0.26–1.24% of total cattle		Loss US$145–885/ranch/year, depending on property and herd size	Peaks in predation in dry season during calving period. 2002–4, 185–240 pumas and jaguars killed. Predation correlated to presence of forest fragments and riparian corridors
Central-western Brazil, cattle ranch[r] (200 km^2)	1998–2002	Puma and jaguar	C	309 cattle killed by predators	0.4% of cattle lost to predators	1.7% lost to disease, malnutrition, accident, and mismanagement	0.3% of total livestock value	Most predation by pumas, proportional to calves available. Some retributive killing of predators occurs

Notes: B, domestic buffalo; C, cattle; Ca, camel; D, dog; Do, donkey; G, goats; H, horse; P, poultry; Pi, pig; S, sheep; Y, yak.

[a] Schiess-meier *et al*. (2007); [b] Patterson *et al*. (2004); [c] Frank (2005), Ogada *et al*. (2003); [d] Kolowski and Holecamp (2006); [e] Mizutani (1999); [f] Rudnai (1979); [g] Bauer (2003), Van Bommel (2003); [h] Stander (1997a, b); [i] Butler (2000); [j] Wang and Macdonald (2006); [k] Saberwal (1990), Srivastav (1997); [l] Bagchi and Mishra (2006), Mishra (1997); [m] Madhusudan (2003); [n] Oli *et al*. (1994); [o] Stahl (2001a, b); [p] Mazzoli *et al*. (2002); [q] Michalski *et al*. (2006); [r] Palmiera (2008).

Figure 6.2 A village headman displays the remains of a cow killed by lions, Tshlotsho Communal Land, adjacent to Hwange National Park. © A.J. Loveridge.

northern Cameroon, levels of lion depredation were inversely correlated with distance to Waza National Park (Van Bommel *et al.* 2007).

Temporal patterns of livestock depredation are often predictable and in many cases vary with levels of available natural prey. In many areas of Africa, livestock depredation peaks during the season when natural prey is most difficult for predators to acquire. This is usually the wet or rainy season, when prey are in better condition and often more widely dispersed (Patterson *et al.* 2004; Bauer and Iongh 2005; Kolowski and Holecamp 2006). Hemson (2003) found that in the Makagadigadi ecosystem, Botswana, livestock-raiding lions killed fewer livestock during periods when migratory wildebeest (*Connochaetes taurinus*) and zebra (*Equus burchelli*) were present in their ranges (Loveridge *et al.*, Chapter 11, this volume). In Laikipia, Kenya, livestock losses to lions and leopards were lower during drought years when wild prey were in poor condition and therefore easier for predators to capture (Ogada *et al.* 2003).

Peaks in livestock predation may also be due to seasonal availability or vulnerability of livestock. In areas with marked seasonal variation in climate, livestock may be in poor condition during the lean season, which may make them easier prey for felids during this period (Hoogesteijn *et al.* 1993; Butler 2000). Neonates of domesticated species are often more susceptible to predation than adults. This vulnerability can result in increased levels of depredation during the lambing or calving period. Peak losses to jaguars and pumas on cattle ranches in southern and central Brazil and central Venezuela often coincide with the calving season (Polisar *et al.* 2003; Michalski *et al.* 2006a; Palmeira *et al.* 2008).

Seasonal changes in husbandry or herding practices may also result in periods of vulnerability for livestock. In France, peak losses (May–November) of sheep to lynx coincided with periods when sheep were kept in fenced fields for the entire 24-hour period, presumably increasing their availability to lynx and therefore the probability of depredation (Stahl *et al.* 2001b). Villages around Bhadra Tiger Reserve, India, experienced lower levels of livestock loss during the harvest season, when it is thought that increased human activity in surrounding fields may also inadvertently provide protection for stock (Madhusudan 2003).

Economic impact of livestock depredation

Impacts of felid depredation on livestock vary, depending on the scale of livestock ownership, husbandry techniques, livestock type, stocking density, and density of predators. Small-scale, subsistence livestock owners are often disproportionately impacted by losses, in part because they may lack resources to provide effective protection for their stock. Loss of even an individual animal to small-scale livestock owners has a proportionally higher impact on herds or flocks than the same loss to owners with more stock. For this reason, Ikeda (2004) found that small-scale yak (*Bos grunniens*) herders in Kanchenjunga Conservation Area, Nepal, were more heavily impacted by snow leopard (*Panthera uncia*) predation on their herds than wealthier, medium- to large-scale owners.

Snow leopard predation in the Spiti Region, India, amounted to losses of between 25% and 52% of annual per capita income for small-scale farmers (Oli *et al.* 1994; Mishra 1997). In Africa, subsistence farmers in Kenya and Zimbabwe lost between 11% and 12% of annual income to lion and leopard depredation (Butler 2000; Ogada *et al.* 2003). Villages on Koyake Group Ranch adjacent to the Maasai Mara Reserve, Kenya, lost approximately US$1890 to lions and US$984 to leopard depredation over 14 months (Kolowski and Holecamp 2006). Tiger depredation on livestock around Bhadra Tiger Reserve, cost villagers 16% of annual income (Madhusudan 2003). Around Jigme Singye Wangchuck National Park, Bhutan, tigers and leopards caused losses of 17% of annual per capita income, however for households actually affected by losses this represented up to 84% of annual cash income (Wang and Macdonald 2006; Wang 2008). These losses are significant impacts on the livelihoods of people, who often have few alternative means of earning a living.

In contrast, large ranches lose relatively less to depredation. Large ranches in Alta Floresta, Brazil, lost relatively fewer cattle to depredation than intermediate-sized ranches, with no ranch losing more than 1.24% of its herd (Michalski *et al.* 2006a). On large-scale ranches in Kenya, it is estimated that the cost of maintaining a lion was US$290 in southern Kenya and US $360 in Laikipia (Patterson *et al.* 2004; Frank *et al.* 2005), while the cost of maintaining a leopard cost ranchers US$200–211/year in Laikipia (Mizutani 1997; Frank *et al.* 2005). The cost of tolerating occasional depredation was considered to be low by these relatively wealthy ranchers (Mizutani 1999). Losses due to depredation by carnivores are often much smaller than losses due to disease, poor husbandry, or theft. On a Laikipia ranch, average loss of stock to predators over a 23-year period was 34 cattle and 66 sheep a year. However, annual losses to disease were 2.5 times higher for cattle and nearly 4 times higher for sheep. Losses attributed to theft were similar to losses to carnivores (Mizutani 1997). In Kweneng district, Botswana, 0.34% of livestock was lost to carnivores each year (although losses to individual owners were higher at 1.0–11% of holdings). However losses to disease, starvation, and accident accounted for 2.8–12.6% of stock (Schiess-Meier *et al.* 2007). On central Brazilian ranches, 0.4% of stock was lost to predators, while 1.7% was lost to disease, malnutrition, and mismanagement (Palmeira *et al.* 2008). In contrast, around Bhadra Tiger Reserve losses of stock to predators were around 4 times higher than losses to disease or starvation (Madhusudan 2003). In Gokwe, Zimbabwe, levels of loss of livestock to predators (~5%) were similar to losses to starvation and disease (Butler 2000).

Attitudes to livestock depredation

People's attitudes to and tolerance of depredations by wild felids vary widely. Antipathy towards carnivores may be a result of historical or cultural attitudes, as well as based on past experiences and personal values. Although generalizations are difficult across widely variable socio-cultural circumstances and it is often difficult to distinguish between the underlying reasons for negative perceptions people hold towards predators, there is some evidence that levels of tolerance livestock owners have for predators are related to magnitude and impact of losses. In the Pantanal, Brazil, tolerance of jaguars by ranchers was at least partially explained by levels of livestock loss (Zimmermann *et al.* 2005a). However, other studies indicate that personal and societal beliefs and perceptions may be equally important in shaping responses to conflict with predators (see Cavalcanti *et al.*, Chapter 17, this volume; Murphy and Macdonald, Chapter 20, this volume).

Economic circumstance may also dictate levels of tolerance. Subsistence farmers in Annapurna Conservation Area, Nepal, who suffered significant losses to depredation held highly negative attitudes towards snow leopards and considered total extermination of these felids the only solution to the problem (Oli *et al.* 1994). However, in the Indian Trans-Himalaya, where people had alternative incomes and lower dependence on livestock, they were more tolerant of snow leopards (Bagchi and Mishra 2006). Subsistence pastoralists on group ranches in Laikipia, Kenya, were more likely to attempt to eradicate predators than neighbouring commercial ranchers (Woodroffe and Frank 2005). In this case, commercial ranchers tolerate losses to large felids both because of a cultural appreciation of wildlife and because wild felids have value to ecotourism operations (Mizutani 1997, 1999). In other areas, people are willing to tolerate moderate losses. In India, villagers often graze livestock on government forest lands and are willing to tolerate moderate loss as the price of access to grazing resources (Srivastav 1997; Karanth and Gopal 2005).

Human injury and fatality caused by felids

Perhaps the most striking relationship people have with felids is that large felids occasionally prey upon, kill, or injure people. Evidence that hominids have always suffered depredation by felids comes from the Swartkrans Caves in South Africa, where the excavated skull of a hominid child bears the tooth marks of a leopard (Brain 1969; de Ruiter and Berger 2000). Carnivores from the families Felidae, Canidae, and Ursidae are all on record as killing people (Quigley and Herrero 2005). Due to the need to overpower and kill their victims, human-killing felids are invariably the largest species, with lions and tigers being the most dangerous and widely publicized (Table 6.2). Famous historical cases of man-eating big cats include the man-eating lions of Tsavo, which, in 1898, are reputed to have killed around 128 construction workers on a railway line being built from Mombasa to Nairobi, Kenya (Patterson *et al.* 2003). Other historical examples include man-eating tigers and leopards in northern India, one of which (a tigress) reputedly killed 436 people before being hunted and shot (Corbett 1946). Boomgaard (2001) reports that in India, in 1875 alone, tigers and leopards killed 1039 people. However, human deaths due to large cats are not a thing of the past. Human deaths and injuries caused by lions are relatively common in central and southern Tanzania, with around 563 Tanzanians killed between 1990 and 2004 (Packer *et al.* 2005b). Similar numbers of people are killed and injured in the Sundarbans of India and Bangladesh, with 294 people killed between 1984 and 2001 in the Indian Sundarbans (Siddiqi and Choudhury 1987; Karanth and Gopal 2005) and 79 people killed between 2002 and 2007 in the Bangladesh Sundarbans (Khan 2007).

Human attitudes to man-eating and their consequences

At a global scale human killing by felids is relatively rare, much rarer than human deaths due to traffic accidents for instance. However, because of the terrible emotional consequences attached to the loss of human life, human killing puts conservationists in a difficult moral dilemma. It makes conservation and protection of big cats difficult for conservationists to justify to local people who suffer losses. Baldus (2004, 2006) gives an example of a man-eating incident in the Mkongo Division, Rufiji district, adjacent to Selous Game Reserve in Tanzania, that illustrates the traumatic disruption to people's lives. In this case, at least 34 people were killed and 10 injured by lions over a period of 20 months, with most incidents occurring in crop-lands south of the Rufiji River, which divides the district. Eventually, almost the entire human population had fled to the northern banks of the river to escape the lions, leaving homes and crops unguarded. Leaving aside the emotional and psychological consequences of living alongside man-eating carnivores, the perceptions and attitudes of people towards wildlife and conservation of large carnivores are seriously impacted. In poor countries, compensation for loss of human life is often grossly inadequate. In Tanzania, compensation for loss of human life to wild animals is around US$30–50 (Baldus 2004). Apparent undervaluation of human life may further compound ambivalent attitudes of local communities towards conservationists and conservation managers, reduce public acceptance of big cat conservation, and undermine conservation efforts. Indiscriminate retaliatory killing of cats often results from incidences of human killing. In the case of the Rufiji maneater, wire snares were set throughout the area to catch the culprit, killing at least eight lions (which may or may not have been maneaters) and probably many other non-target species, before the actual maneater was eventually shot (Baldus 2006). Attempts to eliminate maneaters may impact the viability of local populations of carnivores.

The magnitude of the problem

Table 6.2 provides some examples of accidental killing, injury, or predation on people by felids. Felids from the genus *Panthera* are the most consistent maneaters, although this behaviour is rarely recorded in *P. onca* (Rabinowitz 2005). Pumas (*Puma concolor*) are known to attack and kill people, but incidences are extremely rare, with 10 deaths and 48 injuries throughout the

Table 6.2 Summary of studies on depredation on humans by large felids, showing number and rate of attacks in various areas. The type of attack is classified as either being accidental (A) or as the result of depredation (D).

Location (area km^2)	Dates	Species	Human density	Human injuries	Human deaths	Attacks/year/ 1000 km^2 *	Type attack	Details
Tanzania, Njombe District[a] (2000 km^2)	1944–52	African lion			1000	55.5	A/D	At least 30 lions killed (17 confirmed maneaters). Lions completely extirpated by 1950s
Tanzania, Tenduru District[a] (1560 km^2)	1986	African lion			42	26.9	D	43 lions shot
Tanzania, Lindi District[a] (800 km^2)	1999–2000	African lion			24	15.0	D	Man-eating stopped after seven subadult lions were shot
Tanzania, Mkongo Division, Rufiji District[b] (350 km^2)	August 2002–April 2004	African lion	37.8/km^2	10	35	50	D	1991–2004 at least 58 people killed. Most people killed were guarding crops. 3.5-year-old male maneater killed
Tanzania[c]	1990–2004	African lion		308	563	—	A/D	Attacks most common in districts with low density of natural prey
Mozambique, Niassa Game Reserve[d] (25,000 km^2)	2000–06	African lion	1.0/km^2	17	11	0.19	A/D	~50% attacks occurred in villages, 32% in fields. One lion killed More than 34 people killed and 37 injured by lions 1974–2007
Zambia, Lupande GMA[e] (4800 km^2)	1991	African lion			3	0.62	D	Three lions shot. Lions had old wounds from wire snares, but otherwise healthy. Pride displaced into marginal habitat outside the park by a larger pride
Uganda[f]	1923–94	African lion		69	206	—	A/D	Men attacked more often than women, due to engagement in hunting activity. ~75% of lion attacks fatal. 376 ‘problem’ lions killed
Uganda, Queen Elizabeth National Park[g] (955 km^2)	1990–2000	African lion			32	3.4	A/D	People killed when herding or moving through park at night, and when drunk
Kenya, Tsavo East region[h]	1898	African lion	Low		128	10.0	D	Railway workers targeted by at least two male lions. Work was temporarily halted until lions shot. One lion had cranial deformities and broken teeth
India, Gir Wildlife Sanctuary[i] (1400 km^2)	1978–91	Asiatic lion		165	28	~10.6	D	82% people killed adjacent to reserve, largely by dispersing subadults. Man-eating may have been as a result of drought, low prey density. Similar trend recorded in 1901–04 drought
Uganda[f]	1923–94	Leopard		75	37	—	A/D	One-hundred-and-fourteen ‘problem’ leopards killed over period. Only 32.5% leopard attacks fatal

(*Continued*)

Table 6.2 (Continued)

Location (area km^2)	Dates	Species	Human density	Human injuries	Human deaths	Attacks/year/ 1000 km^2 *	Type attack	Details
India, Gharwhal region[j]	1996–97	Leopard			36	—	D	Women and children (and one drunk) killed by leopard
United States[k]	1890–90	Puma		43	10	—	A/D	30% of cases in vicinity of Vancouver Island, increasing incidence of attacks due to human encroachment into habitat
Russia, Sikhote-Alin Zapovednik[l] (4000 km^2)	1970–2001	Amur tiger	2.8/ km^2	37	14	0.01	A	People often killed/injured when encounter tigers accidentally while hunting. 81.7% of tiger mortality due to people (1992–2001)
India, Kanha Tiger Reserve, Madhya Pradesh[m]	1985–2001	Tiger		25	22	—	A	Attacks thought to be largely accidental
India, Sundarban Tiger Reserve[m] (~4300 km^2)	1985–2001	Tiger		57	294	5.1	D	Tigers appeared to target humans as prey
Bangladesh, Sundarbans[n] (5770 km^2)	1953–83	Tiger			554	3.4	D	Many deaths not reported due to illegal use of the forest
Bangladesh, Sundarbans[o] (5770 km^2)	2002–06	Tiger			79	2.7	D	At least 13 tigers killed in retaliation. Use of dogs to protect/alert forest users being tested
Nepal, Chitwan National Park[p] (750 km^2)	1980–2005	Tiger	52.4/ km^2	Not given	88	0.47	A/D	Tiger attacks have increased over study period, perhaps due to increasing tiger and human population. 50% people killed collecting forest products. Thirty-seven known maneaters (of which 15 killed, 6 captured for zoos, and 4 translocated)
Sumatra[q]	1978–97	Tiger		30	147	—	A/D	Attacks declined in mid-1980s–90s due to declining tiger populations. Twenty-eight man-eating tigers shot/poisoned and 20 translocated. 73% of man-eating tigers were male

Notes: [a] Baldus (2004); [b] Baldus (2006); [c] Packer *et al.* (2005); [d] Begg *et al.* (2007); [e] Yamazaki and Bwalya (1999); [f] Treves and Naughton-Treves (1999); [g] West African working group report (2001); [h] Patterson (2004); [i] Saberwal (1994); [j] Kruuk (2002); [k] Beier (1991); [l] Miquelle (2005); [m] Karanth and Gopal (2005); [n] Siddiqi and Choudhury (1987); [o] Khan (2007); [p] Gurung *et al.* (2006); [q] Nyhus and Tilson (2004b).

* Not calculated for entire countries or regions due to varying/sporadic nature of human killing.

United States and Canada from 1890 to 1990 (Beier 1991). Lions and tigers, the largest living cats, are responsible for most incidents, with episodes of leopards as maneaters being occasionally reported, particularly in northern India (Corbett 1946; Athreya 2006; Athreya *et al.* 2008). Levels of human killing by felids vary widely. Human fatalities to tigers were 0.01 people/1000 km^2/year in the Russian Far East, where human population density was low and interactions with tigers rare and usually accidental (Miquelle *et al.* 2005a). This contrasts strongly with the 50 people/1000 km^2/year killed by lions specifically targeting people as prey items in Rufiji district (Baldus 2004, 2006). By their nature, incidents of human killing by felids tend to be clumped in space and time, so compilations of incidents across large spatial scales or long time periods tend to under-represent the local impact. Likewise, reports that include incidents over a short time-span and localized scale may overstate the global importance of man-eating as a behavioural trait of felids, while at the same time capturing the traumatic impacts such incidents have on human communities living alongside populations of large felids.

It is not always possible to unambiguously distinguish between those incidents that were the result of accidental encounters between a person and large felid and those where the person was viewed as a prey item by the felid in question. Karanth and Gopal (2005) compare the examples of Kahna Tiger Reserve, India, where human deaths are relatively low (but not infrequent), with the Sundarban Tiger Reserve, India, where human killing by tigers is frequent. In Kahna, tigers occasionally come into contact with people in the densely populated areas surrounding the park, leading to occasional human deaths when tigers are disturbed or accidentally encounter people harvesting forest products. However in the Sundarban Tiger Reserve it is clear, both from the high incidence of human deaths and the circumstances of the attacks, that people are the intended prey of some tigers (Table 6.2).

Characterization of human–felid conflict

Situations when livestock depredation or man-eating are more likely to occur fall into a number of broadly defined circumstances. These are scenarios when encounters between people or livestock and felids are more frequent because either felids disperse from existing or emerging populations or when people enter felid habitat. Conflict may also be more likely to occur when levels of natural prey have been depleted. These three situations are discussed in more detail below.

Dispersal of felids into marginal, human-dominated habitat

Ironically, the consequence of successful protection of populations of large felids is that surplus individuals may need to disperse out of habitats or refugia already occupied by conspecifics. If the surrounding areas provide no connectivity to other habitat patches and/or are marginally suitable habitat, already used by people, conflict with human populations in these areas can result.

Examples of this include tigers dispersing into buffer areas surrounding Chitwan National Park, Nepal. This has led to 88 human deaths between 1980 and 2005, with many of the dispersing tigers being old individuals displaced from their ranges (McDougal 1993; Gurung *et al.* 2006a). A similar example of competition for space leading to displacement into areas of human settlement occurred in Lupande Game Management Area, Zambia, where a pride of lions, displaced from South Luangwa National Park by a stronger pride, killed three people (Yamazaki and Bwalya 1999). Conflicts between people and large felids over livestock loss are common on the boundary of many protected areas in Africa and Asia. Large felids disperse into marginal boundary areas and kill livestock. For example, dispersing lions kill livestock around Etosha and Hwange National Parks (Stander 2005b; A.J. Loveridge, personal observation). Similar conflicts between lions, people, and livestock occur in the boundary areas of Gir Forest, India (Saberwal *et al.* 1994). Problems of dispersing carnivores from emerging populations have been experienced in western Europe, where reintroduced European lynx have come into conflict with sheep farmers (Kaczensky 1999).

Human encroachment on felid habitat

People and their livestock enter (legally and illegally) large felid habitat for a number of reasons, the most common being for hunting or extraction of forest products, use of grazing resources, and occasionally for other reasons such as leisure activities, tourism, or as refugees (Johnson *et al.* 2001b; Kruuk 2002). These activities bring people into contact with felids and may result in either accidental killing of people (and/or felids) or provide opportunities for felids to prey upon people or livestock. In the Sundarbans of Bangladesh, there are no permanent settlements in the forest but people use the forest for extraction of forest products both legally and illegally. Here some tigers apparently regularly prey upon people, with fluctuations in numbers of fatalities correlated with periods of high use of the forest (Khan 2007). Similarly, injury or death caused by tigers in the Russian Far East appears to occur when people accidentally encounter tigers while hunting in the forest (Kerley *et al.* 2002; Miquelle *et al.* 2005a). Livestock losses are high in areas where livestock are grazed in or in close proximity to forested habitats with resident felid populations (Wang and Macdonald 2006; Palmeira *et al.* 2008). In Norway, sheep are grazed unprotected in forested habitats, leading to high losses to carnivores (Swenson and Andrén 2005).

Another reason people encroach on felid habitat is to settle in these areas. Nyhus and Tilson (2004b) record the increasing incidence of livestock loss and man-eating by tigers in Sumatra during the 1970s–80s when settlement of the island by transmigrants was encouraged and people encroached on tiger habitat during the massive transformation of the lowland rainforests into oil palm plantations. However, incidents declined during the mid-1980s as tiger habitat was destroyed and tiger populations on the island declined. This pattern of conflict followed by population decline has been closely documented for tigers on the Indonesian islands of Java and Sumatra and in India between 1600 and 1950 by Boomgaard (2001). Similar patterns of depredation by tigers may have occurred across south-east Asia as human populations increased and colonized forest habitats throughout the nineteenth century (McDougal 1993). Rapid fragmentation and conversion of forested habitats to agricultural land in Amazonia bring people and felids into contact and result in high levels of conflict (Michalski *et al.* 2006a). Similarly, the increasing encroachment of human residential development into puma habitat in North America may increase the chances of people encountering pumas and account for the increases in puma attacks on people in the past decade-and-a-half (Cougar Management Guidelines Working Group 2005). People also become vulnerable when circumstance forces them to intrude into big cat habitat, as illustrated by the example of Mozambican refugees, fleeing into South Africa, via the Kruger National Park, being killed and eaten by lions (Kruuk 2002).

Prey depletion causes felids to switch to alternative prey

Evidence from some studies suggests that felids have a preference for natural prey. This may be because availability of domesticated species is not reflected by their numerical abundance, as livestock is often protected and livestock-raiding animals risk higher levels of mortality in human-dominated environments. Eurasian lynx in Norway did not select habitat patches with high livestock densities, but preferred areas with high roe deer density, and preyed predominantly on roe deer, despite the fact that sheep densities were eight times higher (Moa *et al.* 2006; Odden *et al.* 2006). Rabinowitz and Nottingham (1986) found that jaguars prefer natural prey to domestic stock. Hemson (2003) found that lions appeared to 'prefer' wild prey, at least during periods of high wild ungulate abundance. In this case, they killed livestock less than expected based on availability in the good season and in proportion to availability in lean season. Accordingly, it was suggested that maintaining suitable levels of natural prey in protected-area buffer zones may serve to limit depredation on livestock.

A common suggestion in reports of depredation on both livestock and people is that levels of natural prey have been depleted and this may provide an explanation for these behaviours. While this has rarely been quantified, a number of examples appear to support the suggestion and may explain a proportion of the cases. For example, Asiatic lions attacked people around Gir Wildlife Sanctuary, when natural prey densities declined after a prolonged drought (Saberwal *et al.* 1990; Saberwal *et al.* 1994). In the case of man-eating lions in central and southern Tanzania, their natural prey was depleted, forcing the lions to switch to prey (bush pigs, *Potamochoerus porcus*) found in proximity to human settlement.

This may increase the likelihood of encounters with people guarding their crops, potentially leading to man-eating events (Packer *et al.* 2005b). Hoogesteijn (2002) and Crawshaw (2004) found that loss of natural habitat and prey predisposes jaguars to kill livestock. Similarly, losses of domestic stock to snow leopards were high, making up around 58% of snow leopard diet, in areas with low abundance of natural prey (Mishra 1997; Bagchi and Mishra 2006)

Mitigation of conflict

Characteristics of problem animals

Defining a large cat as a problem requiring a solution depends on both where it is and how it is behaving. Linnell *et al.* (1999) distinguish between two kinds of 'problem animal'. The first are animals that are in the 'wrong place' (generally in a wide-scale matrix of habitats, where not all individuals have access to livestock within their home ranges). Individuals dispersing out of protected areas or individuals coming into contact with livestock or human intruders into their habitat might fall into this category. The second 'type' of problem animal is one that kills more livestock (or people) per encounter than do conspecifics. Animals with old injuries or habitual livestock raiders (Stander 1990) could be included in this definition.

There are a number of factors that appear to predispose individual felids to becoming problem animals. Male felids appear to be more likely to become stock raiders (Stander 1990; Cunningham *et al.* 2001; Funston 2001a; Odden *et al.* 2002). Odden *et al.* (2002) found that male Eurasian lynx were not only more likely to kill livestock than females, they were also more likely to make multiple kills during a raiding incident. Specific male behaviours may also be factors that influence whether or not livestock is taken (Linnell *et al.* 1999). Bunnefeld *et al.* (2006) found male lynx were more likely to be found in proximity to human habitation than females with kittens, which may explain both higher male mortality and predisposition to livestock killing by males of this species. Similarly, because felids are markedly sexually dimorphic, with males being larger than females, one might expect that male felids would have a greater size range of livestock available to them, leading to higher rates of predation. However, there are few data to directly support this suggestion.

Higher rates of livestock killing by male felids could also be due to larger male home ranges and wider dispersal distances and therefore an increased likelihood of encountering livestock (Linnell *et al.* 2001). This is certainly the case in lions, a species where subadult males often disperse widely. Patterson *et al.* (2004) found that 66% of stock-raiding lions in southern Kenya were subadult males, while Funston (2001a) records that of 100 livestock-raiding lions in ranch land adjacent to Kgalagadi Transfrontier National Park, 59 were subadult males. In this case, a further 31% of raiders were females with cubs, suggesting the possibility that increased nutritional requirements of lactating females or females feeding dependent young may predispose them to stock raiding.

In a study on the boundary of Etosha National Park, Namibia, Stander (1990) was able to distinguish between habitual and occasional stock-raiding lions. Around 70% of habitual raiders were adult males, while 54% of occasional raiders were subadult individuals. He found that 11 of 12 'occasional' raiders translocated did not reoffend; however, 'habitual' raiders quickly returned to their home ranges and livestock raiding unless they were translocated >100 km.

Infirmity and disability have often been cited as a possible reason for man-eating and livestock depredation and there are a compelling number of examples of maneaters and livestock raiders with old injuries or damaged teeth. This may or may not cause the animal difficulty in capturing or subduing wild prey. For example, over 50% of 17 man-eating tigers from Chitwan had old injuries caused by gunshot wounds, intraspecific fights, or had worn or broken teeth (Gurung *et al.* 2006a). One of the two 'man-eating lions of Tsavo' had deformities of skull and jaw (Peterhans and Gnoske 2001) and the maneater of Rufiji was a 3.5-year-old male with severe abcessation in its lower jaw (Baldus 2006). A number of the notorious man-eating tigers and leopards shot by Corbett were old and had been suffering from disability (Corbett 1946). In post-mortem examinations of Belize jaguars, Rabinowitz (1986) found that of 13 livestock raiders, 10 had old injuries (many caused by old gunshot wounds), while in a sample of 17 non-livestock killers all were healthy. Hoogesteijn *et al.* (1993) report similar trends in Venezuelan jaguars, where 10 of 19 cattle killers suffered from old gunshot injuries. The notion that old age or disability could be the cause of aberrant predation on humans or livestock is appealing. While this appears to be the case in some incidences, it is equally likely that old or infirm individuals lose competitive intraspecific interactions in prime habitats and are thus displaced from core areas and into marginal habitat, where they come into contact with people and livestock. Habitual livestock raiders may have been more likely to have been shot at in the past. Indeed, Patterson *et al.* (2003) found that the skulls of lions killed as problem animals around Tsavo National Park, Kenya, were no more likely to show signs of disability, or tooth breakage than those in museums, which were presumed to have been collected at random.

Solving and mitigating human–felid conflicts

Designing conservation policy and building management capacity

In developed countries where there is a strong commitment to carnivore conservation, compensation schemes have been established to mitigate losses, coupled with selective removal of individual problem predators. In many developing countries, lack of attention by governments or low and delayed compensation discourages livestock owners from participating in similar schemes where these exist. The only perceived solution is to kill the predator, and often many non-target animals are killed in the process. People who fear for their safety or perceive that they are at economic risk will not support conservation efforts.

Conservation practitioners need to design frameworks for implementing a comprehensive response to problem animals. Such frameworks often include creation of professional problem animal response teams and crafting of national policies and protocols for response to conflicts between humans and large felids. Response teams should be professional, well trained and adequately equipped, and have the experience and confidence to handle the range of situations that are encountered and fully understand the consequences of their actions. Many countries have no written policies for dealing with problem carnivores; situations are handled on a case-by-case basis with no guidelines and without the proper staff training, equipment, and resources. A national policy provides a set of protocols that clearly delineate appropriate actions for the various situations likely to be encountered and guide the process of determining the appropriate response options. While every situation is unique, a general protocol that empowers local wildlife managers to make decisions, makes them accountable, and provides response teams with guidelines to follow for choosing the appropriate course of action is a critical first step in dealing quickly and efficiently with problem animals.

Because local support for, or at least tolerance of, large felids is one of the key factors determining the fate of all wild populations, elimination of real or perceived threats is pivotal. The ability to retain a problem animal within the wild population will depend on the abilities of the response personnel, effective liaison with local communities, and the severity of the problem, as well as whether official interventions are adequately directed, resourced, and supported.

Lethal control

Predator reduction or elimination, either through state-sponsored predator control or unregulated killing, has historically been the method of choice for protecting livestock. In this way, large felid populations have been extirpated from large areas of North and South America, Africa, Europe, and Asia. While complete eradication of predators is no longer practised in many areas, reduction of predator numbers or limitation of population growth through lethal control methods may be compatible with management plans that include zonation of predator management.

Where complete eradication of predators is not desirable, lethal control of specific problem animals can be used to deal with livestock raiding or incidents of human death or injury. However, it is sometimes difficult to identify the individual animal responsible and indiscriminate lethal control may kill many non-target individuals or species, particularly if wire snares or poison are used. Control methods that specifically target problem animals, such as toxic collars on vulnerable livestock (Burns *et al.* 1996), use of dogs or skilled trackers to follow problem animals, and shooting or trapping of culprits when they return to recently made kills may provide more focused control of problem individuals (Odden *et al.* 2002; Woodroffe and Frank 2005).

Trophy hunting is sometimes advocated as a means to reduce predator densities or to target specific problem animals. Trophy hunting has been shown to temporarily reduce population densities in some felid species (e.g. puma, Lindzey *et al.* 1992; Eurasian lynx, Herfindal *et al.* 2005a; Stoner *et al.* 2006) depending on the sustained level of the hunting pressure.

Felid populations often recover quickly from population reductions, if there are nearby source populations (e.g. lions, Smuts 1978; Eurasian lynx, Stahl *et al.* [2001a]). Therefore, unless hunting is used on a consistent basis to limit population growth, the benefits to livestock owners are likely to be temporary. Furthermore, vacant home ranges are often filled by dispersing subadults, a demographic group that appear more likely to become problem animals. There is evidence that ranges made vacant may be divided between multiple subadult dispersers resulting (at least temporarily) in increased population density (Laing and Lindzey 1993). This may escalate rather than resolve conflicts. This being the case, sustainable sport hunting which seeks to maintain viable predator populations (and therefore hunting opportunities) may be incompatible with predator reduction (or eradication) in livestock raising areas, unless livestock owners are prepared to tolerate some losses.

In an alternative approach to reducing population density, Anderson (1981) describes proactive culling of potential livestock-killing lions in Hluhluwe-Umfolozi National Park, South Africa. It was found that animals dispersing from the Park and causing livestock losses on surrounding farms were predominantly subadult males (18–42 months old, 59 of 79 problem lions destroyed). Proactive culling of subadult males in the Park resulted in a greatly reduced incidence of livestock loss on surrounding farmland. However, highly targeted interventions such as this may only be practical in intensively managed situations, such as small-fenced reserves where animals are often regularly monitored and individually recognized.

Reducing felid population densities through hunting can provide temporary benefits to livestock owners through reduced depredation. However, use of hunting as a tool to selectively target specific problem animals appears to be a limited and temporary solution (Stahl *et al.* 2001a; Herfindal *et al.* 2005a). Furthermore, identification of the problem animal after the fact is not always straightforward and in situations where tourist hunting opportunities are sold commercially, bogus claims of problem animals need to be guarded against. Lethal control of specific problem animals may be better undertaken by professional problem animal control personnel, rather than the sport-hunting public; the former is likely to be a more targeted and timely response. This is particularly the case when dealing with man-eating predators (Cougar Management Guidelines Working Group 2005).

Where problem animals are part of a rare or endangered population, lethal control may not be the option favoured by conservationists. Nevertheless, particularly in cases where people's lives and livelihoods are at risk, removal of a problem animal by lethal control may be the most expedient and efficient option available. Unsuccessful or ill-conceived interventions or unwillingness on the part of managers or conservationists to deal effectively with problem animals may lead to a perception that the welfare of animals is valued over the lives and livelihoods of local people. In this case, support for conservation efforts may be undermined (Tilson and Nyhus 1998).

Translocation

Other options for removal of problem animals include translocation back into protected areas, to zoos or other protected sites. This intervention has been used in a number of areas, particularly with lions and leopards in Africa, but success is equivocal due to high post-release mortality, extensive movements, and homing behaviour of translocated animals. In Tsumkwe district, eastern Nambia, Stander *et al.* (1997a) translocated six livestock-raiding leopards a total of 12 times to sites 10–135 km from their ranges. All individuals returned within 2 days and killed livestock again in an average of 8.2 (range 1–20) months. Man-eating leopards translocated from Pune district, India, to other forests and reserves continued to reoffend at the new sites (Athreya 2006) and similar behaviour by leopards is described in east Africa (Hamilton 1976). Of 14 stock-raiding male lions (two territorial adults, two non-territorial adults, and 10 subadults) translocated distances of 50–85 km from their home ranges in the Kgalagadi Transfrontier Park, South Africa, all except three subadults returned and killed livestock again within 170 ± 37 days. Most were repeatedly translocated (between two and six times) but either returned or were destroyed while killing livestock (Funston 2001a). Similarly, 40% of the 54 lions translocated from around Etosha National Park returned to their home ranges (Stander 2005b) and for the same reason capture and repatriation of stock-raiding Asiatic lions failed to control livestock raiding around Gir Wildlife Sanctuary, India (Saberwal *et al.* 1990). Similar post-release behaviour has been observed in other large felids. Rabinowitz (1986) reports that two livestock-killing jaguars translocated to Cockscomb Basin National Park, Belize, soon left the park. One continued to kill livestock and was shot after 5 weeks; the other disappeared. An experimental translocation of 14 cougars in New Mexico resulted in 25% mortality within 3 months and, overall, nine animals died during the study period. Two mature males moved back ~477 km to their original ranges in 166 and 469 days. Eight cougars moved >80 km back towards their original range before establishing new home ranges (Ruth *et al.* 1998).

Translocations are most likely to be successful if problem animals are moved long distances across significant landscape barriers to areas with reasonable prey densities and few livestock or people. However, such areas are infrequently available and tend to have already established populations of the species being translocated. Translocation of non-territorial, subadult individuals, which would disperse anyway in a natural setting, appears to be the most likely to be successful. In pumas, translocation was most effective for animals aged 12–24 months and least effective for older individuals (Ruth *et al.* 1998). Based on the results of lion translocation operations in Namibia, Stander (1990) recommended that habitual problem animals be destroyed while occasional raiders, particularly subadults, could be rehabilitated by translocation. While in this case translocation activities reduced the number of lions that had to be destroyed, it required extensive record-keeping and expertise to capture, mark, and re-identify stock raiders. This kind of intensive management is rarely possible in developing countries. Although translocation of problem felids is in general unsuccessful as an intervention, livestock-killing cheetahs (*Acinonyx jubatus*) and problem Amur tigers have been successfully translocated. In Zimbabwe, in the mid-1990s, 21 cheetahs were moved from the south of the country ~450 km to Matusadona National Park in the north, and successfully established a population there (Purchase 1998; Purchase and Vhurumuku 2005). Of four Amur tigers captured and translocated after killing domestic livestock or attacking people in the Russian Far East (Fig. 6.3), two were successful in that they caused no further conflict with people, killed wild prey, and survived their first winter (Goodrich and Miquelle 2005).

As well as translocating problem animals to alternative wild habitats, some problem animals are occasionally moved into captive settings. Five of 37 man-eating tigers from Chitwan were captured and housed in zoos (Gurung *et al.* 2006a), but facilities to house large numbers of such animals are limited and quickly become saturated (e.g. for tigers in Nepal, India, Malaysia, and Indonesia; leopards in India and Sri Lanka; and pumas in North America). Furthermore, wild-caught felids do not generally settle well in captivity (Karanth and Gopal 2005). However, movement into captive settings might be justified for highly endangered species, especially if release of offspring into the wild could enhance reintroduction programmes at a later date.

In addition to limited success, translocation operations are costly and require high levels of technical expertise and logistical support. Funston (2002) estimates that repatriation of lions that broke out of Kgalagadi Transfrontier Park cost in excess of US$218 per lion. Karanth and Gopal (2005) note that tigers are often injured or disabled during translocation, often through damaging their teeth in steel cages that are not adequately designed (Fig. 6.4). On balance, translocation does not appear either particularly successful or a viable tool for management, particularly in countries with limited resources.

Protection of livestock and improved livestock husbandry

Historically, traditional livestock husbandry practices have sought to limit the availability of livestock to predators. However, in areas where predators have been extirpated this knowledge has often been lost.

Figure 6.3 An Amur tiger is released back into the wild by WCS and Inspection Tiger after being rescued from a poacher's snare. © J. Goodrich, Wildlife Conservation Society.

This is the case over much of western Europe, where traditional herding methods such as herd guarding by shepherds and guard dogs and use of secure enclosures have been largely abandoned. This has contributed to conflicts between farmers and re-emerging predator populations such as Eurasian lynx (Kaczensky 1999; Stahl *et al.* 2001b). Where livestock are grazed extensively without supervision or protection, losses to predators are liable to be high. For example, in Norway sheep graze unsupervised in forested habitats, where they suffer high levels of predation to lynx and other carnivores. In contrast, sheep are usually kept in fenced enclosures in Sweden and loss to predators is limited (Odden *et al.* 2002; Swenson and Andrén 2005). Similarly, free-roaming cattle in central and Amazonian Brazil and central Venezuela were subject to relatively high losses to jaguars and pumas (Polisar *et al.* 2003; Michalski *et al.* 2006a; Palmeira *et al.* 2008). By contrast, in southern Brazil, managed flocks and herds suffered lower levels of depredation than those that were free-roaming (Mazzoli *et al.* 2002) and around Jigme Singye Wangchuck National Park, Bhutan, livestock losses were high in villages that did not have stables or corrals, or where livestock was grazed in forest habitats (Wang and Macdonald 2006).

Protective bomas (corrals; Fig. 6.5) and herd guarding (often by young men and boys) are widely used across Africa to reduce livestock losses (Frank *et al.* 2005). Schiess-Meier *et al.* (2007) found that over a 3-year period only three of 2272 livestock-killing incidents (largely by lions and leopards) took place inside bomas in central Botswana. Similarly, Ogada *et al.* (2003) found that livestock losses were reduced with attentive herding, strongly built bomas, and the presence of guard dogs and people. Enclosures constructed of poles or wicker were superior to those built of thorny *Acacia* branches or wire mesh, because they reduced the chances of stock panicking at the sight of a predator and breaking out of the protective enclosure. Similar reductions in depredation resulted from

Figure 6.4 An African leopard trapped in a steel cage trap. Care needs to be taken in design of such traps to avoid damage to the felid's teeth and claws. In general, trapping and translocation of large felids is not a successful or viable management intervention. © A.J. Loveridge.

stone and wire mesh corrals built to protect sheep against snow leopards in Ladakh, India (Jackson *et al.* 2002). However, Kolowski and Holecamp (2006) found that on ranches adjacent to the Masai Mara Reserve, Kenya, leopards preferentially targeted stock in enclosures, particularly those built of poles, but that in general fences, guard dogs, and human activity deterred predators. Guard dogs can be used to protect small livestock (Linnell *et al.* 1996). In Nambia, farmers using livestock-guarding dogs reported a 73% reduction in livestock loss to cheetahs (Marker *et al.* 2005a). Similarly, llamas (*Lama glama*) and domestic buffalo (*Bubalus bubalis*) have been used to protect flocks and herds from pumas and jaguars (Crawshaw 2004; Hoogesteijn and Hoogesteijn 2008). Visual and acoustic repellents, such as bright lights (Sechele and Nzehengwa 2002) and noises (e.g. shotgun blasts, playbacks of bio-acoustic sounds such as sounds of barking dogs; Koehler *et al.* 1990), may be useful in deterring carnivores. However, carnivores habituate quickly to unusual stimuli and it is not clear whether such methods are effective in the long term. Deterrents such as electronic shock collars and taste aversion work reasonably well under experimental conditions, but have not been widely applied to manage livestock raiding (Frank and Woodroffe 2002).

Improved animal husbandry may also reduce livestock losses to predators. Free-roaming cattle in areas such as the Pantanal, Brazil, and Llanos of Venezuela are often in poor condition, increasing their vulnerability to jaguar predation (Quigley and Crawshaw 1992; Rabinowitz 2005). Vulnerable stock such as calves and sheep often require more intensive management. Peak losses to livestock owners often occur during the calving or lambing seasons (Polisar *et al.* 2003; Michalski *et al.* 2006a; Palmeira *et al.* 2008). In areas of South America where puma and jaguar predation is a problem, use of pastures away from forest fragments and close to human habitation is recommended for calving enclosures (Quigley and Crawshaw 1992; Palmeira *et al.* 2008). In addition, management of breeding in cattle herds aimed at limiting birthing to a short period may allow for intensive protection for part of the year and also act to swamp predation (Crawshaw 2004; Palmeira *et al.* 2008). In areas of Norway where sheep are vulnerable to lynx predation, Linnell *et al.* (1996) recommend shifting to cattle farming, as cattle are not preyed upon by lynx.

Losses in productivity often far exceed losses to predators. Cattle raised under semi-wild conditions have extremely low productivity, with pregnancy rates reaching only 40–50% and have very high

Figure 6.5 Livestock kept at night in a secure pole 'bomas' or corrals suffer lower levels of depredation than those left out to graze. © J.E. Hunt.

neonatal mortality and mortality to disease (Hoogesteijn 2002). Improved husbandry not only limits opportunities for predation by felids, but also reduces losses to disease, accident, and theft and therefore also the potential for losses due to poor management to be blamed on felid depredation (Hoogesteijn 2002).

Compensation

Damage compensation schemes provide one approach to mitigate damage caused by carnivores. Payment of compensation for damages has the effect of spreading the economic burden and financial risk between those who must live alongside carnivores and those who wish to see wildlife protected. Compensation for actual carnivore depredation (sometimes called post-damage compensation) is widely used in Norway, Sweden (Swenson and Andrén 2005), and India (Karanth and Gopal 2005). However, while it is clear that compensation may alleviate some of the costs and potentially promote tolerance of damage caused by predators, this approach suffers from some major disadvantages. Verification of damage (to eliminate fraudulent claims and over-estimates of loss) can be time-consuming and expensive and lead to animosity between conservation managers and livestock owners. Compensation without the requirement to demonstrate adoption of approved preventative measures (as is required in Sweden; Swenson and Andrén 2005) can lead to a perverse incentive to neglect livestock. There may also be incentives to wrongly attribute losses due to poor husbandry, accident, or disease to carnivores in order to claim compensation. Furthermore, compensation may act as an agricultural subsidy, providing incentives to expand farming activities with the potential for conversion of additional natural habitat (Nyhus *et al.* 2005). Long time delays between occurrence of damage and payment of compensation and bureaucratic difficulties in making claims may also be problematic. Saberwal (1990) found that 81% of villagers suffering losses to lions around Gir Wildlife Sanctuary did not bother to make claims, due to procedural bureaucracy. Similarly, failure to fully compensate for losses may hamper the effectiveness of compensation schemes. For instance, around Bhadra Tiger Reserve, where significant losses of livestock to tigers occurred, low levels of compensation were received (3% of estimated losses, with only 20% of claims successful; Madhusudan 2003). Corrupt or poorly managed compensation schemes may deepen antipathies and mistrust of managers and conservation efforts. Furthermore, to provide a sustainable long-term solution significant financial resources are required (Miquelle *et al.* 2005a). For instance, in Norway, compensation costs 5 million Euros/year (Linnell and Brøseth 2003). Schemes are rarely feasible in countries with limited conservation resources unless supported by international sponsorship or non-governmental organization (NGO) funding. Unfortunately, few evaluations of post-damage compensation schemes have been undertaken (Nyhus *et al.* 2003; Nyhus *et al.* 2005). It is not known if they are cost-effective and there is little evidence that compensation payments are effective in improving people's tolerance or reducing attempts to eradicate predators. In one of the few analyses available, Hazzah (2006) found that compensation did not improve attitudes to lions or prevent retaliatory killing on Mbirikani Group Ranch, Kenya.

Private insurance schemes to cover costs of carnivore damage have also been attempted; however, in many cases rural farmers are unwilling to cover the relatively expensive premiums required (Nyhus *et al.* 2005). This unwillingness may stem in part from concepts of ownership of wildlife and the perception that compensation is the responsibility of the government or wildlife management institution. However, success of insurance-based schemes has been claimed in improving tolerance of snow leopards in Pakistan, although this scheme was subsidized through ecotourism revenues (Hussain 2003b). Viability of insurance compensation schemes may rely on innovative financial mechanisms such as this.

An alternative to post-damage compensation is conservation performance payments or compensation in advance (Schwerdtner and Gruber 2007; Zabel and Holm-Müller 2008). Such payments are made based on the expectation that damage is likely to occur and is usually linked to successful conservation outcomes (e.g. reduced levels of carnivore mortality and protection of carnivore habitat). In the Sami reindeer (*Rangifer tarandus*) husbandry area of Sweden, conservation performance payments are made on the basis of the number of certified reproductions of lynx and wolverines (*Gulo gulo*). The level of payment is calculated from the expected losses due to each carnivore during its lifetime and amount to around US$29,000 per individual offspring (Zabel and Holm-Müller 2008). The advantage is that payments are made regardless of damage; therefore, livestock protection efforts are not distorted in that there is no disincentive to protect livestock from predators. Similarly, because payment is made on the basis of expected damage, there is no time-lag in payments, which eliminates uncertainty and promotes transparency and trust. Nevertheless, transaction costs remain high, as carnivore populations still need to be independently assessed and agreement needs to be reached over levels of payment. In common with post-damage compensation payments, viability of schemes requires significant funds over extended periods. Care needs to be taken that payments do not encourage new immigration into the area or subsidize increases or expansion in livestock ownership.

Alternative livelihoods, benefit sharing, and stakeholder participation

Antipathy towards carnivores may be a result of historical and cultural attitudes as well as based on experiences of loss or damage. These attitudes may be difficult to change or ameliorate. However, there are examples where stakeholder attitudes to both conservation and presence of carnivores have been improved by access to alternative revenue or livelihood choices, benefit sharing, and local participation.

Co-management and stakeholder participation appear to improve attitudes towards conservation efforts. In reindeer husbandry areas of Sweden, carnivore surveys jointly undertaken by reindeer owners and conservation managers reduced conflicts over estimates and encouraged transparency and co-operation (Swenson and Andrén 2005). Participation of local communities in prioritization, conservation planning, and implementation enhanced local protection of snow leopards as well as improving the feasibility and sense of ownership of livestock protection initiatives in Hemis National Park, India (Jackson and Wangchuk 2004). Likewise, inclusion of local landowners in conservation meetings in the Pantanal, Brazil, improved perceptions of jaguar conservation efforts and engendered ownership of processes and decision-making (Rabinowitz 2005).

Oli *et al.* (1994) found that villagers who benefited from ecotourism were more willing to tolerate the presence of snow leopards. There is a higher tolerance of this species in areas where people are less dependent on livestock and have access to alternative livelihoods (Mishra *et al.* 2003; Bagchi and

Mishra 2006). In Namibia, ranchers who raise livestock using ecologically sound management techniques gained access to niche markets and premium prices for 'predator friendly beef'. Benefits from initiatives such as this may increase tolerance for cheetahs and other predators on ranch land (Marker and Dickman 2005; Marker *et al.*, Chapter 15, this volume). On commercial ranches in Laikipia that benefit from ecotourism or have wealthy foreign investors, landowners are more tolerant of depredation on livestock than are ranches that gain nothing from tourism. Similarly, traditional pastoralists do not tolerate predators at all: lions dispersing into these areas from commercial ranches are promptly poisoned (L. Frank, unpublished data). However, residents of communal areas that hope to profit from future wildlife tourism profess greater tolerance than those who are not planning conservation-based enterprises (Romañach *et al.* 2007). Stander *et al.* (1997a) describe an innovative ecotourism venture that combined use of traditional skills, revenue generation, and ecotourism. In the Kaudom area of Namibia, local people experienced losses of livestock to leopards. A pilot ecotourism venture took advantage of the local Ju/Hoansi people's exceptional tracking and bush-craft skills. They were able to track and find leopards for paying tourists in 92.9% of attempts. The two villages engaged in leopard-tracking tourism earned N$10,005 (~US$2140) in a year. This revenue amounted to 12 times the value of livestock lost to leopard over the entire district.

However, in many cases claims that revenue from ecotourism or tourist hunting alleviate conflict and improve attitudes towards conservation are almost entirely untested (Walpole and Thouless 2005). Incentive schemes are often heavily subsidized by external bodies (Mishra *et al.* 2003; Miquelle *et al.* 2005a) and internalizing costs to ensure sustainability is often challenging with uncertain outcomes. Wildlife revenues appear to work well in incentivizing protection of habitat when landowners are reasonably wealthy and where owners have land tenure and control over resource access and use (Bond *et al.* 2004). However, there is less incentive where resource ownership is centrally or communally controlled, land tenure is less secure, or where livelihood alternatives are absent. Ecotourism revenues can be mismanaged, misappropriated, or subverted by elites within communities, which may undermine the effectiveness of initiatives (Walpole and Thouless 2005). Furthermore, revenues earned through ecotourism may be enthusiastically accepted in the short term, but in the long term may be invested in development or expansion of agriculture (Murombedzi 1999). The key here is that benefits from ecotourism schemes need to be clearly linked to the need for sustainable conservation of ecosystems and natural resources.

Zonation of land use

Not all landowners or communities will tolerate predators, and certain livestock management practices and human-dominated or urban landscapes are incompatible with the presence of predatory felids. Thus geographic differentiation of land-use areas, ranging from complete protection, through areas where predators are tolerated and population management and/or utilization occurs to areas where predators are not tolerated, seems a sensible way of focusing conservation efforts and resources, while at the same time recognizing the importance of people's livelihoods.

The goals of zoning conservation of carnivores are to conserve viable populations of predators and to minimize, or at least mitigate, conflicts with people (Linnell *et al.* 2005). Limiting the interface between people and large carnivores can serve to reduce the areas where conflict occurs. This allows prioritization of mitigation efforts and efficient use of conservation resources in these areas. Enforced zoning schemes can be used to prohibit certain human activities, though forms of resource use or extraction that do not result in habitat modification such as fishing, regulated logging, regulated harvest of ungulates, leisure activities, and tourism may be compatible with conservation of felids (particularly those that do not pose a threat to human life). In some cases, the presence of charismatic felids may increase the desirability of an area as a leisure or tourist destination. The conservation of smaller species may be compatible with land uses that exclude the presence of larger predators. For instance, small felids pose no threat to livestock and coexistence may be possible if no habitat modification occurs.

Many southern and east African protected area networks have been designed to incorporate areas of strict protection (national parks or reserves), and areas where wildlife can be sustainably utilized (often through trophy hunting). In addition, community-managed areas (known as game management or wildlife management areas) have been established in some countries (e.g. Botswana, Zambia, Zimbabwe, and Namibia), where wildlife is tolerated and utilized for the benefit of communities (Lewis and Alpert 1997; Hulme and Murphree 2001). Utilization areas are often set up as buffer zones between national parks and agricultural areas. Zonation in other regions is often less clear-cut. For instance, many areas of Europe are already inhabited by people. In these areas, initiatives to prioritize conservation of predators have not always met with success and have in some instances been abandoned (Linnell *et al.* 2005).

Establishment of reserves without adequate separation (e.g. with fences or natural barriers such as large rivers) of people and wildlife has led to intense conflict on the peripheries. Clearly, management of conflict in areas where predators and people are expected to coexist makes zonation of predator conservation a challenge. In areas where people and wildlife share the same landscape, it would appear that coexistence is most likely to be successful under circumstances where disadvantages due to conflicts are outweighed by benefits of tolerating wildlife.

Some conservation practitioners have advocated separation of people and predators as the most viable solution, particularly in areas where human densities are high. Nyhus and Tilson (2004b) reported that Way-Kambas Reserve in Sumatra successfully protected tigers and eliminated conflicts with the surrounding human communities. This was largely because the reserve excluded people and reserve borders are largely formed by rivers that created boundaries seldom transgressed by either tigers or people. Karanth and Madhusudan (1997) and Karanth and Gopal (2005) support the voluntary relocation of people from the vicinity of tiger reserves to reduce conflicts and protect tiger populations. Such voluntary relocations to make space for expanding tiger populations have met with success in the Terai Arc Landscape, Nepal (Seidensticker *et al.*, Chapter 12, this volume).

In the western United States, zonation provides the framework for management of puma populations (Laundre and Clark 2003; Stoner *et al.* 2006). Here, subpopulations within the meta-population to be managed are designated as 'source' or 'sink' populations. Sport-hunting opportunities are provided in the sink areas, while source areas are protected from utilization. Management using a landscape of sources and sinks to provide both adequate protection for the species and opportunity for utilization may prove to be an effective strategy for conserving wide-ranging predators. However, understanding of a species' behavioural ecology is crucial. Predators, particularly large felids, are wide ranging and minimum protected-area sizes need to take this into account if edge effects are not to have negative impacts on population viability (Woodroffe and Ginsburg 1998; Loveridge *et al.*, Chapter 11, this volume).

Exploitation of felid populations through trade and utilization

Human cultures have always prized products derived from large carnivores. Traditional cultures and fashionable society value felid furs to advertize personal status and trade in furs has had significant impacts on felid populations. Felid products are used to 'cure' illness, to ward off misadventure, and to bring good luck. For instance, tiger bone as an ingredient of Traditional Asian Medicines (TAM) is thought to cure rheumatism, weakness, and paralysis (Mills and Jackson 1994). In south-western Nigeria, serval (*Leptailurus serval*) flesh and tongue are believed to cure leprosy and rheumatism, while leopard skin is used to treat snake bites (Sodeinde and Soewa 1999). There are recent reports about a growing illegal market for tiger meat as an exotic cuisine (Damania *et al.* 2008). Utilization of felids through trophy or sport hunting can provide motivations for conserving habitats and wildlife populations. We discuss the exploitation of felid populations through trophy hunting, fur trade, and for traditional medicines, and the potential threats and impacts these have on conservation of these populations.

Trophy hunting

Hunting large felids for sport has occurred for thousands of years, with records of lion hunts going back to the time of Pharaoh Amenhoteb III in 1400 BC and medieval tapestries depicting European nobles hunting lions and other felids (Guggisberg 1962). Indian and Nepalese royalty hunted extensively and records show the astounding number of tigers they killed for sport. The Maharajah of Surgujah is reputed to have shot 1150 tigers in his lifetime, followed closely by the Mararaj of Gwalior and his guests, who accounted for 900. British civil servants and military officers on leave hunted frequently, with some individuals credited with killing hundreds of tigers. The British Royal Family did their bit when visiting India and Nepal, with King George V killing 39 tigers in 11 days in 1911 and the Prince of Wales shooting seven over a 4-day period in 1922 (Sankhala 1978; Mountford 1981). During the 1960s, demand for tiger-hunting trophies accelerated as tiger populations dwindled and as it became apparent that hunting tigers would become officially prohibited (Sunquist and Sunquist 2002).

In colonial Africa, carnivores were viewed as vermin and killed wherever possible (Fig. 6.6); this hunting was, at least partially, motivated by 'the thrill of the chase' particularly among the colonial elite hunting in the more remote areas of east and southern Africa. Early hunting expeditions to the Serengeti typically killed large numbers (sometimes hundreds) of lions (Turner 1987), often in the mistaken belief that removal of carnivores would 'protect' ungulate populations, thereby providing improved opportunities for hunting. Along with loss of habitat, decline in prey populations and predator eradication initiatives, trophy hunting may have contributed to historical declines in some large felid populations. Indeed, I.R. Pocock (1939: p. 219) wrote: 'In all parts of the world occupied by Europeans where lions occur, the disappearance of lions is merely a question of time.'

Historically, hunting of carnivores was often characterized by lack of any controls or regulation (at least for the elite) and in some senses was indistinguishable, in motivation and effect, from predator eradication activities that were prevalent at the time. Contemporary trophy hunting (also known as sport, tourist, or recreational hunting) tends to be more restrained and harvests are often strictly regulated where competent management authorities are in place. In addition, trophy hunters are often concerned about and promote conservation values (Lindsey *et al.* 2006). Felids are popular quarry species where trophy hunting is legal, and large charismatic species attract large trophy fees (usually collected by government or management authorities). Trophy hunters pay premium prices to hunting guides or operators to hunt large felids. For example, puma hunts in New Mexico cost US$2000–3000 per hunt

Figure 6.6 Early settlers in east and southern Africa eliminated many of the larger predators from the land they colonized. (Photograph courtesy of National Archives of Zimbabwe.)

(Logan *et al.* 2004) and before hunting was officially banned, hunters paid US$11,200 to hunt a snow leopard in Mongolia (Anonymous 1989). In Botswana, in 2006, tourist hunters paid US$140,000 to hunt a male lion, with US$100,000 of that going to the community in the district it was killed (J. Rann, personal communication), although lion hunting has since been banned in Botswana (Fig. 6.7).

Revenues generated from trophy hunting can benefit local communities (if explicitly channelled) and provide employment for specialists that provide hunting-related services (e.g. hunting guides and taxidermists). This can provide justification and support for conservation of populations of felids and other species at local and national scales. Revenues can also be used to manage populations, protect habitats, or to compensate livestock owners for losses, thereby improving levels of tolerance and acceptance of predators (Loveridge *et al.* 2007b). Furthermore, habitats are often set aside and protected for trophy-hunting activities. For instance, in addition to national parks and protected areas, 1.4 million km^2 are used and, to varying degrees, protected for trophy hunting in sub-Saharan Africa (Lindsey *et al.* 2007). Crawshaw (2004) argues that allowing trophy hunting of problem jaguars may provide incentives for landowners to manage habitats to encourage self-sustaining populations of this species.

Figure 6.7 The skin and skull of a trophy-hunted lion is displayed in a hunter's skinning shed. © J.E. Hunt.

However, trophy hunting can have demographic consequences for felid populations. Adult males are often targeted and high male turnover can result in high levels of infanticide, which may in turn reduce reproductive success within the population. If severe, this can lead to population decline (Greene *et al.* 1998; Whitman *et al.* 2004). Gross (2008) found that heavy hunting of male pumas in Washington, DC, United States, led to severe perturbation including few resident adult males, influx of immigrants from surrounding home ranges (leading to widely varying puma densities), increased infanticide, and more puma–human conflicts. However, felid populations are relatively resilient to moderate levels of harvest (e.g. 10% of adult male lions; Greene *et al.* 1998), recovering quickly if immigration from surrounding habitat or nearby populations is possible. Smuts (1978) found that populations of African lions in Kruger National Park, South Africa, recovered within 17 months (largely through immigration) in areas that had been experimentally depopulated by culling of 129 lions from three experimental areas. However, the social disruption caused by removals persisted beyond the 17-month experimental period (Smuts 1978). Relatively rapid recoveries occurred in lion population in Hwange National Park, after trophy hunting had been suspended in surrounding hunting concessions (Loveridge *et al.*, Chapter 11, this volume). However, Lindzey *et al.* (1992) found that recovery of an experimentally manipulated puma population did not occur within 2 years when 28% of harvestable age animals were removed from the population.

Trophy hunters and hunting operators (outfitters) do not always behave ethically or responsibly. There are often strong financial incentives to overexploit hunted resources, and hunting activities often occur in remote areas with little or no official oversight. For example, Spong *et al.* (2000a) found that in a sample of trophy-hunted leopards ($n = 77$) shot in Selous Game Reserve, Tanzania, 28.6% were female despite hunting quotas for leopards being officially restricted to males. In this case, poor supervision of hunting activities allowed hunters to shoot females and potentially overexploit this resource. Effective regulation of hunting activities to ensure compliance with hunting quotas and regulations is often required to ensure that trophy hunting of felids and other species remains sustainable.

Despite examples of overexploitation (Spong *et al.* 2000a; Yamazaki 1996; Loveridge *et al.* 2007c), trophy hunting appears to have relatively low impacts on felid populations if there are large, protected source populations and if management practices and quota setting are informed and regulated by carefully monitoring the hunted and source populations. Protection of habitat and the potential for benefit streams for local communities and economies might outweigh behavioural and demographic impacts on felid populations, although clearly these should not be ignored if the overall goal is to sustain the target populations over the long term.

Exploitation of felids for their fur

Felid furs are often beautifully patterned and humans have, for millennia, used furs for manufacturing clothing, adornments, and household decoration. The wearing of felid furs often confers status and is a symbol of the wealth and power of the owner. Masai *morans* (young warriors) who have killed a lion wear lion mane headdresses to signify their courage. The fashion of wearing fur coats made of spotted cat pelts gained immense popularity among the wealthy of western Europe and the United States in the 1960s and 1970s. The resulting overexploitation of some wild populations of spotted cats prompted the enactment of the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) in 1975. Recently, tiger and leopard skins and clothing made from skins have become popular among the newly wealthy in China and Tibet (EIA 2004; EIA-WPSI 2006), driving an unprecedented onslaught of poaching in source countries, especially India and Nepal.

Demand for felid skins has driven widespread exploitation and sometimes extirpation of felid populations. One famous example is the extraction of over 80,000 ocelot and 15,000 jaguar skins from the Brazilian Amazon in the early to mid-1960s for the fashion fur trade (Smith 1976). This massive export of skins prompted the Brazilian government to ban export of wildcat skins in 1967. Myers (1973) estimates that the fur trade was worth around US$30 million in the late 1960s and early 1970s. The demand for spotted cat skins was so high that, in Neotropical South America, hunting for cat skins became a way of life for many local people in remote areas (Payan and Trujillo 2006). Hunting provided a lucrative livelihood and extraction of forest products was well organized through intermediaries and middlemen. During this period, in Columbia, a *Tigrillada* (cat hunter) was typically paid around US$130–350 for a jaguar skin, which became worth US$520 to a middleman once it reached Bogotá, before being eventually sold to a New York furrier for US$2500 and made into a coat worth US$20,000 in a New York Boutique (Nowell and Jackson 1996; Payan and Trujillo 2006).

The quantities of spotted cat skins extracted from South American range states from the 1960s to 1980s are staggering. In 1970, 140,000 ocelot and margay (*Leopardus weidii*) skins were traded in US markets, while 84,493 oncilla (*L. tigrinus*) skins were traded in 1983 alone (McMahan 1986). Between 1976 and 1979, 341,588 Geoffroy's cat (*Oncifelis geoffroyi*) and 78,239 Pampas cat (*O. colocolo*) skins were exported from Buenos Aires (Mares and Ojeda 1984). Old World spotted cats were also popular with furriers, including the leopard, snow leopard, clouded leopard, tiger, and cheetah (Nowell and Jackson 1996). Nine thousand, one hundred and sixty-two leopard skins were exported from Uganda between 1924 and 1960, leading to special protection of the species in that country in 1960 (Treves and Naughton-Treves 1999). Seventeen thousand, four hundred and ninety leopard skins were exported to the United States in 1968 and 1969 and demand for leopard skins was estimated at around 50,000 skins per year in the early 1970s. Furthermore, because leopards were often hunted on an unmanaged and often illegal basis by tribesmen or subsistence hunters (Fig. 6.8), many skins were rejected by middleman traders as being damaged or poorly cured, implying that losses to wild populations were even higher than figures suggest (Myers 1976).

Figure 6.8 Demand for leopard skins led to widespread hunting of the species, with at least some of the illegal trade in skins supplied through a network of illegal trappers and hunters. (Photograph courtesy of National Archives of Zimbabwe.)

Concern over the magnitude of trade in spotted cats and its potential impact on wild populations (Koford 1973; Myers 1973, 1976; Theile 2003) led to moratoria on trade in some species (Nowell and Jackson 1996). Protective legislation was introduced in many range states and CITES legislation restricting trade in many of the spotted cats, particularly the larger species, was enacted in 1975. The European Union (EU) banned all imports of Latin American cats in 1986 and many of the smaller Latin American cats were placed on CITES Appendix I between 1989 and 1992. Although trade in spotted cats still exists (1000 spotted cat skins were seized in Argentina in 1990; Anonymous 1992), trade restrictions have led to a steady decline in trade (Figs. 6.9 and 6.10). For instance, 30,000 margay skins were exported in 1977 (most from Paraguay to western Europe) but trade in this species was reduced to 138 skins by 1985 (McMahan 1986). Similarly, in 1969, at the peak of the trade in spotted cats, it is estimated that 61,000 leopards were killed for their skins, with no more than 6000 killed by 1988 (Martin and de Meulenaer 1988).

However, the fur trade shifted to other species as substitutes, such as the bobcat (*Lynx rufus*), and leopard cat (*Prionailurus bengalensis*), with both demand and prices increasing dramatically in the late 1970s (Fuller *et al.* 1985; McMahan 1986). As a result of high demand, prices for bobcat pelts rose from US$15 a pelt in 1973 to a high of US$164 a pelt in 1978 (Fuller *et al.* 1985), with resultant harvests reducing populations of this species to dangerously low numbers (Hornocker 2008). The modern fashion fur trade relies on pelts from the Canadian and Eurasian lynx (*L. canadensis* and *L. lynx*), bobcat, and the leopard cat (Nowell and Jackson 1996). Canadian lynx furs have been exploited since the 1700s (Elton and Nicholson 1942). Although trapping is currently well regulated, populations may be unsustainably trapped during cyclic population declines (Poole 1994; Slough and Mowat 1996). Bobcat populations currently appear stable, suggesting that fur harvests are biologically sustainable; however, very high volumes of leopard cat skins exported by China (over 200,000 skins in 1987) have been cause for concern (Nowell and Jackson 1996; Sunquist and Sunquist 2002).

While markets in Europe and the United States have declined due to strict legislation and negative sentiments about use of furs in fashion items, other markets have opened up, particularly in a more prosperous eastern Europe and an increasingly wealthy China. The illegal trade in tiger and Asiatic leopard (*P. pardus fusca*) skins appears to have been increasing throughout Indochina since the 1990s (Rabinowitz 1999; EIA 2004; Nowell and Xu 2007). Poaching of wildlife for the fur trade appears to be well organized and lucrative. Increasing demand for skins is thought to be driven by wealthy Han Chinese who value felid

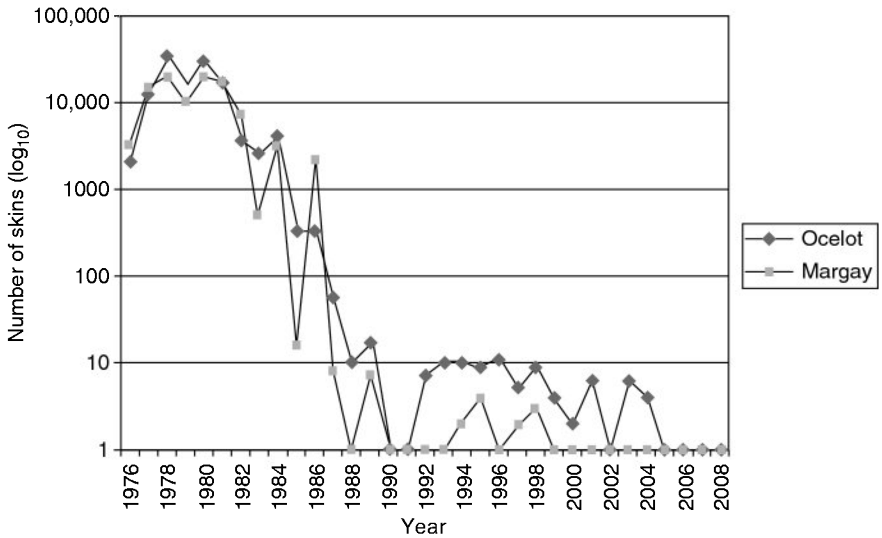

Figure 6.9 Trends in international trade in skins of two species of Neotropical spotted cat (ocelot and margay). Reduction in demand and imposition of trade bans by the EU and CITES in 1986 and 1989, respectively has limited the trade. These data are likely to be incomplete, as only official exports are recorded, but, unless seized, illegal ones are not. (Source of data: CITES–UNEP.)

Figure 6.10 Jaguar skin from the Venezuelan Chaco. © A. Taber.

(particularly tiger) skins for the prestige of owning an expensive and exotic item and because they are thought to bring good luck. There also appears to be a market in curios purchased by Western tourists (EIA 2004). Use of tiger and leopard skins in traditional Tibetan '*chubas*' (ceremonial gowns) became increasingly fashionable through the 1990s as a result of growing wealth among urban Tibetans. Skins are also used at traditional ceremonies such as weddings, and a ceremonial tent made of 108 whole tiger skins was seen in 2006 at the Litang Horse Festival, in the Chinese Province of Sichuan (EIA-WPSI 2006).

However, concerns raised by environmental groups have seen this use decline markedly (EIA-WPSI 2006;

Banks and Wright 2007). Symbolic burning of tiger and leopard skin robes occurred in Tibet in 2006 following condemnation of use of wildlife products by the Dalai Lama. This may have resulted in both greatly reduced demand and prices for these items (Huggler 2006), although Chinese officials, who discourage the Dalai Lama's influence, ordered Tibetans to continue to wear their traditional apparel, including clothing adorned with tiger and leopard skin. Trade in tiger and leopard skins remains highly lucrative. A trader in India is able to sell a tiger skin for US$1500 (having paid a local poacher in the region of US$15), while the same skin sells in China for US$16,000. A consignment of 31 tiger, 581 leopard, and 788 otter skins, seized in Tibet in 2003 while being smuggled from India to China, was thought to be worth an astonishing US$1.2 million (EIA 2004; EIA-WPSI 2006).

Though legislation is in place to protect tigers and most other Asian cats, wildlife agencies frequently lack the resources for effective enforcement of the laws. Poaching penalties are harsh, but the likelihood of apprehension remains low and that of conviction even lower (Damania *et al.* 2008). Hunting of tigers for trade is widespread and trade is thought to be more prevalent now than in previous decades. This may exacerbate declines due to logging, habitat loss, and expansion of human populations (Rabinowitz 1999). The 783 tiger and 2766 leopard skins seized in India between1994 and 2006 are thought to represent a fraction of skins on the illegal market, suggesting that the extent of illegal poaching of felids for their skins may be extensive. EIA-WPSI (2006) consider the illegal trade in tiger skins between India, Nepal, and China to be a substantial threat that may have driven recent declines in Indian tiger populations. Having depleted many tiger populations, commercial poachers have turned to other Asian big cats: Asian lions, leopards, snow leopards, and clouded leopards. Having poached tiger populations in Cambodia, Myanmar, and Thailand, commercial poachers have intensified and focused their efforts on Malaysia and now tigers and other wildlife there are under heavy pressure (Damania *et al.* 2008).

Poaching is usually undertaken by skilled local hunters using reusable steel traps, cable snares, or poisoned bait. The cost of poaching a wild tiger is small, unlikely to exceed US$ 100–200, even taking into account opportunity costs of time and expected penalties. A large number of poachers operate under near open-access conditions. The carcass is sold to traders who capture the bulk of the profit by smuggling tiger parts into urban centres of East Asia. All parts of the tiger can be traded with a total retail value in the region of US $10,000–70,000 (Damania *et al.* 2008).

In India, wildlife poaching and related crime is well organized, with extensive networks of suppliers and buyers (Kumar and Wright 1999). The highly profitable nature of the trade and lack of serious policing and judicial disincentives in many range states do little to curtail the trade in Asian cat skins. Although countries like China appear to be enforcing wildlife trade laws more vigorously (Nowell and Xu 2007), fighting wildlife crime is not a major priority in many South-east Asian range states. Agencies charged with environmental protection are often under-resourced and clear policies and political will are often non-existent. Punishment for trading in wildlife products provides little deterrent, and fines for possession of illegal skins are paltry in relation to the value of a single skin on the illegal market. Traders can adjust their margins to accommodate change in judicial pressure, competition, or demand at the retail end of the market and thus frustrate initiatives to diminish incentives to poach (Bulte and Damania 2005). Furthermore, prosecution of wildlife criminals is frequently delayed and rates of conviction are low. For instance, in India, which has a well developed institutional structure for conservation, only 14 convictions have been achieved in 748 cases of confiscation of wildcat skins (EIA 2004). The trade also spans a number of countries (Fig. 6.11), with sources in India, Indonesia, Laos, Thailand, Myanmar, and Vietnam and markets on the Chinese

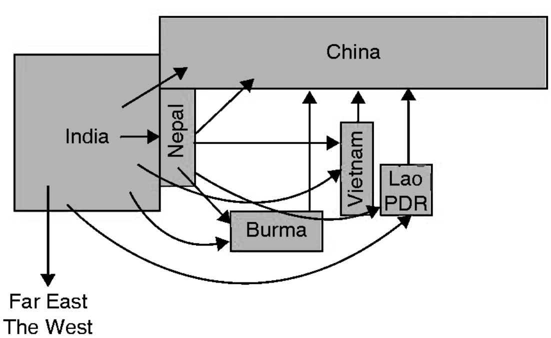

Figure 6.11 Illegal trade routes for tiger and leopard skins in Asia. (From Baker *et al.* 2006.)

Mainland, Hong Kong, Taiwan, South Korea, and Japan (Li and Wang 1999; Rabinowitz 1999; Baker *et al.* 2006; EIA-WPSI 2006). Control of this illegal trade requires regional and international commitment and transnational cooperation to tackle transboundary trade.

Use of felids in TAM

Conservationists have been increasingly concerned by trade in wildlife products used in Traditional Asian Medicines (TAM) and remedies. Trade in tiger parts and derivatives has been banned around the world for more than a decade and law-abiding practitioners of Traditional Chinese Medicine (TCM) now use alternatives. Yet, the illegal trade continues. The World Federation of Chinese Medicine Societies (WCMS) has declared that tiger parts are not necessary for human health care and that alternatives are plentiful, affordable, and effective. Poachers continue to kill tigers to satisfy a stubborn demand for tiger bones to make health tonics. While loss of habitat may be the major long-term cause of declines of species such as the tiger, across East and South Asia, poaching of tigers and other felids is the important short-term threat to survival of populations (Mills and Jackson 1994; Hemley and Mills 1999; Rabinowitz 1999; Wingard and Zahler 2006; Damania *et al.* 2008). Bones of other felids (clouded leopards, leopards, snow leopards, and lions) are also used as substitutes for tiger bone and the trade may therefore also impact populations of these species (Wingard and Zahler 2006). The major Chinese investment and engagement in Africa may present a new threat to lion populations, as it is nearly impossible to distinguish between lion and tiger bones.

Trade in tiger parts, particularly bone, reached epic proportions in the 1980s and 1990s, decimating wild populations. For example, South Korean customs records from before South Korea acceded to CITES show that 8951 kg of tiger bone were imported from 1970 to 1993, with around half the bone derived from Indonesia. There has since been a concerted effort to control trade in tiger bone. Efforts have focused on international legislation, raising public awareness, dialogue with TAM specialists and users, and a commitment among TAM specialists to search for alternative medicinal products to replace the use of tiger bone (Hemley and Mills 1999). Domestic and international trade in tiger parts is banned in most countries and some countries like South Korea have virtually eliminated the trade through vigorous prosecution. However, the residual trade has proved much harder to control and eliminate. In regions where enforcement and prosecution have been indifferent, tiger products are still obtainable. In a survey of Yunnan Province, China, Li and Wang (1999) found felid parts to be widely available and similar surveys in Sumatra, Indonesia, found little evidence of declining trade in tiger products, despite extensive efforts to raise awareness (Shepherd and Magnus 2004; Ng and Nemora 2007). However, a recent survey of 518 TAM outlets in China found that only 2.5% carried tiger bone products, suggesting that use may be declining in Mainland China (Nowell and Xu 2007). However, because few data are available from before the 1993 ban, quantification of declining trade is difficult.

There are a number of captive bred populations of tigers, leading to the suggestion that products from 'farmed' tigers could be sold commercially to replace products from the wild. However, the consensus among conservationists is that trade in farmed tigers is likely to stimulate demand and provide the means to launder tiger parts derived from poaching, thereby reversing recent successes in reducing the magnitude of the trade (Hemley and Mills 1999; Gratwicke *et al.* 2007; Nowell and Xu 2007). A recent survey measured attitude towards consumption of tigers in six Chinese urban areas. Of 1880 respondents, 43% had consumed some product alleged to contain tiger products. The results indicated that while urban Chinese people are generally supportive of tiger conservation, there is a huge residual demand for tiger products which could potentially be filled by supplying parts from both wild and farmed tigers if the ban on trade in tiger parts is lifted in China (Gratwicke *et al.* 2008b).

Management of trade and exploitation

The Convention on Biological Diversity (CBD) explicitly recognizes sustainable utilization as a component of conservation of biodiversity and ecosystems (Convention on Biological Diversity 2003). The key to sustainable utilization is that use is responsibly managed and that populations are well monitored to ensure that overexploitation does not occur. However, sustainable utilization is often challenging in situations where there is an inequity in distribution of wealth, weak environmental controls, apathetic national and international legislation, and endemic corruption. Smith *et al.* (2003) found that poor governance, corruption, institutional failure, and social and economic upheaval reduce the likelihood of successful conservation of biodiversity. Overexploitation of wildlife resources often goes hand in hand with uncontrolled habitat conversion (Rabinowitz 1999; Nyhus and Tilson 2004a) and poaching of tigers is often linked to existing human–tiger conflicts, with illegal trade being a by-product of conflict (Kumar and Wright 1999; Johnson *et al.* 2006a). Household economic and food insecurity has been shown to predispose people to illegal use of wildlife. In Zambia, improvement in food security reduced poaching and improved commitment to conservation initiatives (Lewis and Jackson 2005).

Sustainable use of felid populations is clearly possible, as evidenced by a sustainable fur trade in North America (Nowell and Jackson 1996). Key to this has been clear policies, effectively applied legislation, and population monitoring and research. These conditions do not exist in the developing world. Trophy hunting has the potential to be highly sustainable given its capacity for high revenue generation and protection of habitats and populations, provided institutional mechanisms are in place to assure its sustainability. It is also governed by international trade agreements, such as CITES, and through government conservation institutions. Also, sport hunters as a demographic grouping are often active in promoting the conservation of the ecosystems they utilize.

Trade in cat furs and the medicinal trade are less easy to control through international trade restrictions. However, systematic commercial hunting for spotted cat skins has seen a steady decline since the imposition of trade restrictions by the United States, EU, and CITES, indicating that international action can have a positive impact. In addition, consumer awareness and sensibilities surrounding use of cat skins greatly reduced demand and aided the demise of spotted cat trade in the Southern Hemisphere. This example may provide a template for controlling overexploitation of other felid populations, such as trade in Asian cat skins. If the downturn in demand seen for spotted cat skins is to be experienced for the Asian cats, a combination of international and national legislations, consumer awareness, and education and engagement with the TAM industry is a first step in seeking to reduce trade in tiger and other felid bones (Hemley and Mills 1999). The importance of both strictly applied legislation and public education is borne out by the decline in use of felid parts in China, where a concerted effort has been made to eliminate trade and raise awareness (Nowell and Xu 2007).

Control of unsustainable trade in felids appears to depend on strong international controls, the willingness and capacity of governments to implement and police conservation policy and provide disincentives to poaching and illegal trade, and the action of local and international conservation lobbyists and pressure groups. Consumer awareness and education and engagement of stakeholders also appear crucial in reducing or at least curtailing demand for felid products.

Conclusions

The world's ever-growing human population has dramatically degraded most natural ecosystems, causing an extinction event unprecedented in the last 65 million years. Large carnivores are the first species to disappear under the onslaught of human populations (Woodroffe 2001), having been eradicated first from Europe, then decimated by Europeans in North America and more recently in Asia and Africa. Today they persist only in large protected areas and in diminishing numbers in the least human-dominated of natural ecosystems. Barring a sudden and unlikely reversal in human population trend, the march towards expanding economies, or attitudes towards the natural world, persistence of the larger felid species into the next century is a challenge we must all face. The outlook is dire for those who cherish wild cats. Smaller felids that do not conflict with man and are not commercially overexploited are likely to persist only where their habitats are not deforested, overgrazed, or converted to agriculture.

The threats posed by humans to large felids are soluble if people sharing their landscapes have either economic or ethical incentives to preserve them. Like the ban on ivory trading that saved African elephant populations to date, the successful ban on trade in tropical spotted cat skins shows that conservationists can have a significant influence on international measures to preserve economically important species in the developing world. Moreover, demand was reduced when wearing spotted cat furs became socially unacceptable, suggesting that moral pressures on society may be equally effective. However, the continuing decline of two of the most iconic felid species, the tiger (Chapron *et al.* 2008a; Damania *et al.* 2008) and lion (IUCN 2006b), provide a lens through which to view the challenges facing conservationists when underlying conservation threats are not readily subject to legal or social measures.

Although conflict with people, habitat loss, and reduced availability of wild prey are a significant problem for some tiger populations, poaching to supply markets for traditional medicine and skins in Asia poses the greatest threat to the species (Dinerstein *et al.* 2007; Chapron *et al.* 2008a). It is unlikely that the trend will reverse in the absence of effective legal measures to control trade and the sort of moral pressure that made wearing of spotted cat furs unpopular in the West. Intransigent superstition and the dictates of fashion, increasing prosperity, and cultural inertia in Asia, ensure an increasing and unprecedented demand by consumers for skins and bones. Meanwhile, pervasive corruption at all levels defeats legal sanctions. Demand for tiger products can only be reduced through education about alternatives and development of a sense of personal responsibility for nature and wildlife and improved practice in wildlife conservation. Unless consumers can be convinced that wildlife is more valuable roaming in forests than dismantled in apothecaries and upscale homes, tigers and many other species seem doomed. However, it seems unlikely that such a shift in attitude can be imposed by Western conservationists. It will only accompany change in deeply held cultural values, a slow process compared to the short time left for some wild tiger populations.

In contrast, African lions are declining largely due to continuing poverty and cultural modernization in Africa. At a time when increasing numbers of impoverished people and their livestock are devastating semiarid ecosystems, pastoralist people are also abandoning traditional and effective methods of livestock protection in favour of agricultural pesticides that allow them to eliminate entire populations of predators at very low cost. Again, a major factor is ineffective law enforcement due to lack of resources, lack of interest, and pervasive corruption. However, while tigers are poached because their parts are so valuable, lions are speared, trapped, and poisoned largely because they lack any economic value to the humans that share their habitat. While both tourism and sport hunting are highly lucrative and that money could potentially be used to offset the costs of livestock depredation, very little profit from tourism and hunting reaches the people whose precious livestock are killed by lions. Thus lions and other wild animals are nothing but an expensive nuisance to rural people, with the inevitable result that they are being eradicated. Unless pastoralists and other rural Africans can earn significant income from wildlife, there is no reason for them to conserve it. International fascination with these charismatic animals gives them potentially great economic value. However, this value is largely inaccessible to the rural people who bear the brunt of living alongside large carnivores. Big cats will survive in the wild only if they become more valuable alive than dead to the people who share their landscapes.

Conflict between people and large carnivores has been a consistent theme throughout human history. As human populations burgeon, increasing pressure on remaining populations of carnivores will occur. Our understanding of the processes and patterns of use and conflict may determine our ability to mitigate overexploitation and conflicts and ensure survival of predatory carnivores in a world increasingly dominated by the needs and aspirations of the human species.