CHAPTER 7

People and wild musteloids

Lauren A. Harrington, Jorgelina Marino, and Carolyn M. King

Trapper with North American river otter. Photo from G. Schmitt, schittent.com

Introduction

Musteloids are perhaps unique among the Carnivora in their diversity; accordingly, they encounter, or cause, a diversity of potential problems, both perceived and real, when they interact with people, their activities, and their interests. Among the terrestrial musteloids, only one qualifies for the title of 'large carnivore', and the smallest is no bigger than an elongated mouse; but all are powerful predators for their size. Many have extensive global ranges; others are endemic to a single island. The musteloids living at colder, high latitudes have dense, high quality fur (Kitchener et al., Chapter 3, this volume) and have been exploited for their pelts for centuries. Many have been farmed for their fur, and have been spread across the world by the commercial interests of people. Some have adapted to live among humans (Macdonald et al., Chapter 1, this volume) and are considered a nuisance or kept as pets; others are among the most endangered mammals in the world.

In this chapter we focus on two key areas: contemporary exploitation of wild musteloids, and conflict between people and native musteloids. For each, we draw on the published literature and publicly available harvest and trade statistics, and summarize species information available on the IUCN Red List of Threatened Species, to provide a broad overview of the issues, their impact (for people and the musteloids themselves), and their management. We start by outlining briefly the historic impact people have had on musteloids, stemming from the early attitudes of people towards them, partly as a reminder of how things can go very wrong, but also to provide context for what follows.

Harrington, L. A., Marino, J., and King, C. M., *People and wild musteloids.* In*: Biology and Conservation of Musteloids*. Edited by David W. Macdonald, Chris Newman, and Lauren A. Harrington: Oxford University Press (2017).
 DOI 10.1093/oso/9780198759805.003.0007

We do not refer to farmed or invasive species here (but see Fraser et al., Chapter 16, this volume, and King et al., Chapter 10, this volume), nor do we discuss disease-related issues (but see Newman and Byrne, Chapter 9, this volume).

Historic impact of people on musteloids

Early attitudes and perceptions

Throughout history, people's attitudes towards, and perceptions of, musteloids have been as varied as the species they pertain to. And it is these attitudes and perceptions that have, at least in part, influenced the way in which people have both used and managed these small carnivores, with consequences for the musteloids that ranged from negligible to devastating.

In early Europe, small mustelids were much appreciated for their elegance and for their services in killing rodents. Many of the traditional vernacular names for the least weasel (*Mustela nivalis*) in Italy and Spain are complimentary, such as *donina* (little lady or little graceful one), *bonuca-mona-muca* (pretty little one), and *comadreja* (little godmother).

Birks and Kitchener (1999), in contrast, refer to early mentions of the polecat (*Mustela putorius*) in British literature as 'wholly derogatory', citing Shakespeare to illustrate the use of the name 'polecat' for 'vile' people and prostitutes: 'Out of my doore you Witch, you Ragge, you Baggage, you poulcat, you Runnion, out, out' (The Merry Wives of Windsor, Act 4, Scene 2). The name 'polecat' may be derived from the early French expression *poule chat* (chicken-cat) in reference to the animal's perceived pest status and both its Latin name *Mustela putorius* (which translates into 'foul smelling musk bearer') and its Old English name 'foulmart' refer to its smell (although it is not the only musteloid to produce a pungent odour).

Over-exploitation and the historic fur trade

In medieval Europe, the furs of northern musteloids were symbols of luxury and status, reserved for the upper ranks of the nobility. The best quality sable (*Martes zibellina*) skins from the Barguzin valley in southeast Siberia, still known as Imperial sable, were worn only by the Russian royal family. Trade in Russian furs goes back to the tenth century, and by the seventeenth to eighteenth centuries Russia was the world's largest supplier of furs. Hundreds of thousands of ermine (stoats in white winter fur, *Mustela erminea*), marten (*Martes* sp.), and sable were trapped in Siberia each year (Forsyth 1992), and their pelts sold at the annual autumn Leipzig trade fair, the largest fur market in Europe.

The maritime fur trade peaked between the 1780s and the early 1800s, when the Chinese would pay very high prices for the thick sea otter pelts brought to Canton from the cold waters of the Northwest Pacific. Boston-based ship-owners sailed round Cape Horn with American trade goods, exchanged them for pelts collected by Native American tribes living along the coast, sold the pelts in Canton, and used the proceeds to load cargoes of tea to take back to Boston (Gibson 1992). There was frantic competition between American, Russian, Spanish, and British fur traders for this 'soft gold' and between 1804 and 1837 American traders alone sold 158,070 sea otter skins (an average of 4790 a year), although numbers were dwindling towards the end of this period (Kruuk 2006; Figure 7.1). By 1840, sea otters numbers were depleted across their range, and hunting sea otters was eventually banned in 1911 (by which time there were an estimated 2000 animals left, Kenyon 1969; Estes et al., Chapter 23, this volume). Sea mink (*Neovison macrodon*) that formerly ranged along the coast of Canada and eastern North America (and

Figure 7.1 Sea otter skin, Unalaska, 1892. Photographer Stefan Claesson. Gulf of Maine Cod Project, NOAA National Marine Sanctuaries; Courtesy of National Archives.

whose pelts were also apparently sought after through the 1800s) were not so lucky: the last sea mink was recorded in 1894 (Campbell 1988).

In the 1920s and 1930s, fur came to epitomize glamour, and an average of 700,000 weasels and almost 200,000 mink (*Neovison vison*) were harvested annually in Canada (Proulx 2000). Mink fur was the most coveted at the time (and mink furs still dominate the market today, although most are now farmed). Wolverines (*Gulo gulo*, subject to trapping and persecution since the mid-nineteenth century, see Banci 1994; Robitaille 2000) were harvested only in low numbers (< 1000 annually) through most of the twentieth century, but harvests of fisher (*Pekania*, formerly *Martes, pennanti*) and marten (*Martes americana*) continued to increase, peaking at 15,000 and almost 200,000, respectively, in the 1980s (Proulx 2000). Meanwhile, in South America, 38,000 'otter' (marine [*Lontra felina*] and/or southern river otter [*L. provocax*]) skins were exported from Chile between 1910 and 1954 (an average of over 800 per year, Iriarte and Jaksic 1986), and similar numbers of giant otter (*Pteronura brasiliensis*) skins (24,282 over 25 years between 1946 and 1971) from Peru (Schenck 1997 cited in Kruuk 2006). Both were surpassed by the export of Neotropical otter (*L. longicaudis*) skins (also from Peru) in the late 1960s to early 1970s, at a rate of 8000 or more per year (Smith 1981). During the same period, almost five million raccoons (*Procyon lotor*) were harvested in the US, many of which were shipped to Europe and sold as imitation otter or mink fur (Zeveloff 2002).

Fur trapping never made a substantial long-term difference to populations of ermine (King and Powell 2007), but other furbearing musteloids (e.g. Dunstone 1993; see also Maran et al., Chapter 17, this volume) were all affected to various extents. Marten (Helgen and Reid 2016a), wolverine (Banci 1994; McKelvey et al. 2014), North American river otters (*L. canadensis*, Bricker et al. in press), and South American Neotropical otters (Rheingantz and Trinca 2015) had all been exterminated from parts of their former range by the early twentieth century, and, like the sea otter before them, giant otters came close to extinction in the early 1970s, before international trade restrictions came into force in 1975 (Groenendijk et al. 2015a; see also Groenendijk et al., Chapter 22, this volume).

Local persecution and the case of traditional gamekeeping in Britain

Farmers and hunters worldwide have persecuted predators for centuries (Reynolds and Tapper 1996; McDonald and Murphy 2000) and we illustrate the effects on musteloids with the case of traditional gamekeeping in Britain in the late nineteenth to early twentieth century. The policy in Britain at the time was to increase the gamebird population available for shooting on privately owned estates by indiscriminate suppression of all predatory mammals and birds (Figure 7.2), carried out by over 25,000 gamekeepers equipped with guns, poisons, and steel traps (Tapper et al. 1982).

Several species of carnivores and raptors, which had previously been widespread, disappeared from most of Britain. The pine marten (*M. martes*) vanished from England and most of Scotland during the first half of the nineteenth century, and the polecat during the second half (Yalden 1999). Step (1921, cited in Packer and Birks 1999) reflects the prevailing attitudes in the country at the time: writing with reference to the rarity of the polecat, Step expresses gratitude to gamekeepers for their 'unremitting vigilance', and remarks: 'there is no more destructive beast among our native Carnivora'.

Figure 7.2 Predatory birds (crows and owls) and mammals (stoats and weasels) hanging on a game keepers' gibbet, c.1968. © P. Morris

At the same time, huge numbers of foxes (*Vulpes vulpes*), badgers (*Meles meles*), stoats, and weasels were also killed, but none of them suffered the same widespread, catastrophic population decreases as polecats and pine martens. Otters (*Lutra lutra*) succumbed later to the effects of agricultural pesticides (Strachan and Jefferies 1996), and although badgers were protected from gamekeepers to some extent in their underground setts they were locally decimated (Skinner et al. 1991). Stoats and weasels were (and still are) extremely resistant to control (King and Powell 2007): artificial mortality imposed by gamekeepers often merely substitutes for, rather than adds to, their already high natural mortality.

Many of the species that declined due to overexploitation and persecution in the nineteenth and early twentieth centuries have now recovered in Europe and North America (some quicker than others), due to protective legislation and reduced persecution. Eurasian otters have, similarly, recolonized much of their previous range due to improved water quality and reduced pesticide use (Crawford 2010). Ironically, these 'conservation successes' sometimes result in reigniting old conflicts (e.g. between otters and fish farms in Europe, Kranz 1994; Ring et al. 2008), or in new conflicts with activities developed while they were rare (e.g. sea otters [*Enhydra lutris*] and the commercial and recreational red abalone [large edible sea snail, *Haliotis* spp.] fisheries off the Californian coast, Fanshawe et al. 2003). We return to each of these examples in the third section of this chapter where we review the myriad of human–musteloid conflict scenarios that arise in the modern world. First, we discuss the continuing, contemporary exploitation of musteloids, for fur and other uses.

Contemporary exploitation of musteloids

When considering wildlife exploitation, musteloids are probably not the first taxa to spring to mind. Elephant and rhino poaching, and trade in tiger skins and pangolin scales, are instead all well known and widely reported. Indeed, Sonricker Hansen et al.'s (2012) web crawling surveillance system, designed to monitor official and unofficial reports of illegally traded wildlife (at international, national, and local levels), listed only six reports involving musteloids among a total of 705 in the year April 2015–2016. However, there is some concern that smaller and lower profile species, such as the musteloids, and particularly the otters, have been overlooked and their exploitation gone largely unnoticed (IOSF [International Otter Survival Fund] 2014). Certainly, trade in rare otter skins is extensive in some parts of the world; they are smuggled alongside tiger parts, and ten or more otter skins are found for every one tiger skin (IOSF 2014).

According to the IUCN Red List of Threatened Species (version 2016.3, www.iucnredlist.org, hereafter the IUCN Red List), at least 69 musteloid species are hunted or trapped (legally and illegally, to some extent), predominantly for the fur trade (over 50 species) but also for food (bushmeat), use in traditional medicine, and, increasingly, the pet trade (Figure 7.3, Table 7.1). We describe each of these four main uses, and how they might affect musteloids below. Only three musteloids (common raccoon, European polecat, and lesser grison [*Galictis cuja*]) are listed as being hunted for sport, but Japanese marten (*M. melampus*) are also considered a 'game species' (Abramov et al. 2015), as are wolverines in Russia (Abramov 2016a), and European badgers are 'heavily hunted' (for unspecified purpose) in Finland (Kranz et al. 2016). Crab-eating raccoons (*Procyon cancrivorus*) are reportedly subject to 'target practice' (Reid et al. 2016b), and annual trade records collated under the Convention of International Trade in Endangered Species of Wild Fauna and Flora (trade.cites.org, hereafter CITES) show over 100 honey badger (*Mellivora capensis*) skins, bones, or 'trophies' exported from African range states (predominantly to the US) in 2012 and 2014. Among 'other' uses, pygmy spotted skunks (*Spilogale pygmaea*) are stuffed and sold as souvenirs to tourists in Mexico (Helgen et al. 2016a). Subsistence hunting of sea otters (for food, furs, and the creation of handicrafts) by Alaskan Native Americans is a special case permitted under the Marine Mammal Protection Act, which prohibits commercial harvest of sea otters, and under which non-natives would face hefty fines and jail time (Golden 2008).

Relatively few (13) of the terrestrial musteloids subject to hunting and trapping are 'threatened' species (classified by the IUCN as Vulnerable, Endangered, or Critically Endangered, Table 7.1), and, for those that are, the overall impact of hunting and trapping is generally considered to be low. One exception is the Endangered red panda (*Ailurus fulgens*), for which poaching and illegal trade seems to be increasing (and is a major threat in Myanmar, Glatston et al. 2015). For some non-threatened species, hunting and trapping is a potential concern because it is considered to be unsustainable (e.g. pine marten in parts of its range, Herrero et al. 2016), or because the species is not well protected (e.g. Siberian weasel [*Mustela sibirica*], Abramov et al. 2016b).

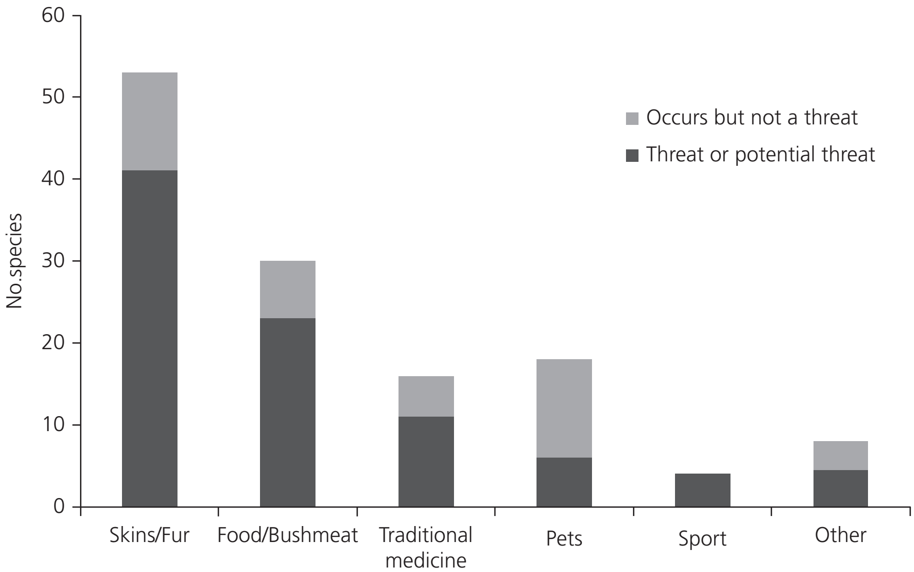

Figure 7.3 Number of musteloid species hunted or trapped for various uses (legally and illegally), based on threats and uses (i.e. occurs but is not considered a threat) as listed in IUCN Red List species entries (version 2014.5), with additional information on use from furbearer harvest data and CITES trade records (see Table 7.1).

Table 7.1 Musteloid species that are killed for their uses (exploitation) or in response to conflict, indicating those species for which 'Hunting & trapping' for 'Intentional Use' (where the species is the target) (IU) or for 'Persecution/Control' (P/C) is classified as a threat, according to the IUCN Red list of Threatened Species (version 2016.3). Y = Yes (i.e. it is a threat), N = No, and indicates a species that is known to be killed (for either reason)* but for which killing is not currently categorised under the list of threats. – indicates a species that is not known to be either Used or Persecuted/Controlled. Particular uses that are referred to within the IUCN Red List as a threat are checked in bold; L=local use, N=national, I=international. Note, however, that information on particular uses of musteloids is not always readily available and actual impacts on species status are often unknown – these data should therefore be considered only as an indication of the types of uses and the potential range of species that may be affected. IUCN status: CE = Critically Endangered, E = Endangered, V = Vulnerable, NT = Near Threatened, LC = Least Concern. Population trends: D=declining, S=stable, I=increasing, U=unknown. CITES: I, II, III indicates the CITES appendix under which the species is listed (and, for appendix III species, the listing country if only a single country, see cites.org).

SPECIES	**HUNTING & TRAPPING IS A THREAT**		**USES**						**SPECIES STATUS**		**CITES LISTING**
	IU	**P/C**	**Fur**	**Food**	**TM**	**Pets**	**Sport**	**Other**	**IUCN status**	**Pop. trend**	
TERRESTRIAL MUSTELOIDS											
Red panda *Ailurus fulgens*	Y	–	✓	✓	✓	✓			E	D	I
Hog badgers *Arctonyx* spp.	Y	–	✓	**✓ L**					LC/V	D/S	
Olingos *Bassaricyon* spp.	N	–				✓ L		✓	LC/NT	D	III
Ringtail *Bassaricus astutus*	Y	–	**✓ L, N**						LC	U	
Cacomistle *Bassaricus sumichrasti*	Y	–	**✓ L, N**	**✓ L**					LC	U	III (Costa Rica)

continued

Table 7.1 *Continued*

SPECIES	HUNTING & TRAPPING IS A THREAT		USES						SPECIES STATUS		CITES LISTING
	IU	P/C	Fur	Food	TM	Pets	Sport	Other	IUCN status	Pop. trend	
American hog-nosed skunk *Conepatus leuconotus*	N	–	✓ L, N, I						LC	D	II
Molina's hog-nosed skunk *Conepatus chinga*	Y	–	**✓ L, N**						LC	D	II
Striped hog-nosed skunk *Conepatus semistriatus*	Y	–	**✓ L, N, I**						LC	U	II
Humboldt's hog-nosed skunk *Conepatus humboldtii*	Y	–	**✓ L, N**			**✓ L, N**			LC	S	II
Tayra *Eira Barbara*	N	Y	✓ L			? I			LC	D	III
Lesser grison *Galictis cuja*	Y	Y				**✓ L**	✓		LC	U	
Greater grison *Galictis vittata*	Y	–			✓	**✓ L**			LC	S	III
Wolverine *Gulo gulo*	Y	Y	**✓ I**				✓		LC (V Europe)	D	
Saharan striped polecat *Ictonyx libycus*	Y	–			✓				LC	U	
Striped polecat *Ictonyx striatus*	–	Y							LC	S	
Patagonian weasel *Lyncodon patagonicus*	–	Y							LC	U	
American marten *Martes americana*	Y	–	**✓ N, I**						LC	D	
Yellow-throated marten *Martes flavigula*	N	–	✓ L, N						LC	D	III
Stone marten *Martes foina*	Y	N	✓						LC	S	III (India)
Nilgiri marten *Martes gwatkinsii*	–	Y							V	S	
European pine marten *Martes martes*	Y	Y	**✓ L, N**						LC	S	
Japanese marten *Martes melampus*	N	–					✓		LC	S	
Fisher *Pekania (Martes) pennanti*	Y	–	**✓ L, N, I**						LC	U	
Sable *Martes zibellina*	Y	–	**✓ N, I**						LC	I	
Asian badger *Meles leucurus*	Y	–	✓	✓					LC	U	
European badger *Meles meles*	Y	Y		✓	✓		✓		LC	S	

continued

Table 7.1 *Continued*

SPECIES	HUNTING & TRAPPING IS A THREAT		USES						SPECIES STATUS		CITES LISTING
	IU	P/C	Fur	Food	TM	Pets	Sport	Other	IUCN status	Pop. trend	
Japanese badger *Meles anakuma*	–	–							LC	D	
Honey badger *Mellivora capensis*	Y	Y		✓	✓ **L, N**	? I	? I		LC	D	III (Botswana)
Bornean ferret badger *Melogale everetti*	Y	Y						✓[1]	E	D	
Small-toothed ferret badger *Melogale moschata*	Y	–	✓ L	✓ **L**	✓ L				LC	US	
Javan ferret badger *Melogale orientalis*	Y	–							LC	U	
Large-toothed ferret badger *Melogale personata*	Y	–	?	✓ **L**	✓ L				LC	U	
Hooded skunk *Mephitis macroura*	Y	N	✓ I	✓ **L, N**				✓[1]	LC	I	
Striped skunk *Mephitis mephitis*	Y	N	✓ **L, N, I**						LC	S	
Amazon weasel *Mustela africana*	–	–							LC	U	
Altai mountain weasel *Mustela altaica*	Y	–	✓ L, N		✓				NT	D	III (India)
Stoat *Mustela erminea*	N	N	✓ I						LC	S	III (India)
Steppe polecat *Mustela eversmanii*	Y	Y	✓ **L, N**						LC	D	
Colombian weasel *Mustela felipei*	–	N							V	D	
Long-tailed weasel *Mustela frenata*	N	–	✓ L						LC	S	
Yellow-bellied weasel *Mustela kathiah*	N	N	✓ L, N						LC	S	III
European mink *Mustela lutreola*	Y	–	✓ L, N						CE	D	
Indonesian mountain weasel *Mustela lutreolina*	N	–							LC	S	
Least weasel *Mustela nivalis*	–	Y	✓	✓	✓				LC	S	
Malay weasel *Mustela nudipes*	N	–		✓	✓				LC	D	
European polecat *Mustela putorius*	Y	Y	✓ **L, N**				✓ **L, N**		LC	D	

continued

Table 7.1 *Continued*

SPECIES	**HUNTING & TRAPPING IS A THREAT**		**USES**						**SPECIES STATUS**		**CITES LISTING**
	IU	**P/C**	**Fur**	**Food**	**TM**	**Pets**	**Sport**	**Other**	**IUCN status**	**Pop. trend**	
Siberian weasel *Mustela siberica*	N	N	✓ **I**	✓					LC	S	
Back-striped weasel *Mustela strigidorsa*	N	–	✓ **L, N**		✓ **L**				LC	S	
Indonesian stink badger *Mydaus javanensis*	N	–		✓ L	✓			✓[2]	LC	S	
Palawan stink badger *Mydaus marchei*	Y	–		✓ **L**		✓			LC	S	
South American coati *Nasua nasua*	Y	–	✓ **L, N**	✓ **L**		? I			LC	D	III (Uruguay)
White-nosed coati *Nasua narica*	Y	–	✓ **L, N, I**	✓ **L**		? I			LC	D	III (Honduras)
Mountain coati *Nasuella* spp.	Y	–	✓ L, N	✓ L		✓		✓[1]	E/NT	D	
American mink *Neovison vison*	Y	N	✓ **N, I**						LC	S	
African striped weasel *Poecilogale albinucha*	Y	–			✓				LC	U	
Kinkajou *Potos flavus*	Y	–	✓ **L, N**	✓ **L**		✓ **L, N, I**			LC	D	III (Honduras)
Crab-eating raccoon *Procyon cancrivorus*	Y	–	✓ **L, N**			✓ **L, N**		✓[3]	LC	D	
Common raccoon *Procyon lotor*	Y	N	✓ **L, N, I**				✓ **L, N**		LC	I	
Pygmy raccoon *Procyon pygmaeus*	N	–				✓ L			CE	D	
Pygmy spotted skunk *Spilogale pygmaea*	Y	–						✓[4] **L, N**	V	D	
Eastern spotted skunk *Spilogale putorius*	N	–	✓ L, N						V	D	
Western spotted skunk *Spilogale gracilis*	Y	–	✓ **L, N**						LC	D	
American badger *Taxidea taxus*	Y	Y	✓ L, N						LC	D	
SEMI-AQUATIC MUSTELOIDS (OTTERS)											
African clawless otter *Aonyx capensis*	Y	Y	✓ **L**	✓ **L**	✓ **L**				NT	D	I
Asian small-clawed otter *Aonyx cinereus*	Y	Y	✓ **L**	✓ **L, N**		N			V	D	II
Congo clawless otter *Aonyx congicus*	Y	–	✓ **L**	✓ **L**	✓ **L**			✓[5] **L**	NT	D	I

continued

Table 7.1 *Continued*

SPECIES	HUNTING & TRAPPING IS A THREAT		USES						SPECIES STATUS		CITES LISTING
	IU	P/C	Fur	Food	TM	Pets	Sport	Other	IUCN status	Pop. trend	
Sea otter *Enhyda lutris*	Y	Y	✓ **L**						E	D	I & II
Spotted-necked otter *Hydrictis maculicollis*	Y	Y	✓ **L**	✓ **L**	✓ L				NT	D	II
North American river otter *Lontra canadensis*	Y	–	✓ **L, N**						LC	S	II
Marine otter *Lontra felina*	Y	Y	✓ **L, N**						E	D	I
Neotropical otter *Lontra longicaudis*	Y	Y	✓ **L**			✓ L			NT	D	I
Southern river otter *Lontra provocax*	Y	–	✓ **L, N**						E	D	I
Eurasian otter *Lutra lutra*	Y	Y	✓ **L, N, I**	✓ **L**					NT	D	I
Hairy-nosed otter *Lutra sumatrana*	Y	–	✓ **L, N**	✓ **L, N**	✓ **L, N**	? N			E	D	II
Smooth-coated otter *Lutrogale perspicillata*	Y	Y	✓ **L, N**	✓ **L**	✓ **L**	N			V	D	II
Giant otter *Pternura brasiliensis*	Y	Y	✓ **L**	✓ **L**		✓			E	D	I

*in this case, we consider 'Uses' as listed by the IUCN Red List but also furbearer harvest data and CITES trade records as evidence of use, and details from the IUCN Red List text as well as information from the scientific literature as evidence of persecution/control.
[1]= not specified; [2]= perfume (Java); [3]= target practice; [4]= stuffed and sold as souvenirs (Mexico); [5]= handicrafts and jewellery

The fur trade

Despite protests by animal rights campaigners, poor public image, and a slump in the market in the 1990s, the fur trade continues to play a significant role in the global economy in the twenty-first century. According to news reports in 2015 (e.g. Orange 2014, Euse 2015), fur seems to be firmly back in fashion. Figures published by the International Fur Federation suggest that the fur trade was worth over 40 billion dollars to the global economy in 2012/2013 (as much as Wi-Fi, www.wearefur.com), and wild fur represents 15–20% of that trade.

As in the nineteenth century, most wild fur-trapping is still carried out in North America and Russia, but China is now the biggest global importer of both wild and farmed fur (Orange 2014, and is the largest producer of farmed furs). In 2014, almost 3.5 million pelts were harvested in the USA and almost half (47.4%) of those were musteloids (National Furbearer Harvest Statistics Database, www.fishwildlife.org).

North America

Wolverine skins are currently the most valuable of North American musteloids, and of the US furbearers were beaten only by Arctic timber wolf (*Canis lupus lycaon*) and bobcat (*Lynx rufus*) skins in the 2016 auctions (Table 7.2). Raccoon skins are among the least valuable of the North American musteloids (partly because they are considered 'too bulky and heavy' for the Chinese fashion market, Jansen 2015), but over a million of them are still trapped every year (Table 7.2, Figure 7.4; see Zeveloff, Chapter 27, this volume).

Fur harvests in North America are now regulated, primarily by state wildlife agencies, through restricted species-specific trapping seasons, annual quotas ('bag limits') and permitted methods (White et al. 2015), and are carried out by an estimated 176,573 state-licensed trappers (over half of which are based in the Midwest, AFWA [Association of Fish and Wildlife Agencies] 2015).

Table 7.2 Musteloids harvested for their fur in North America. Other North American furbearers, and Barguzinsky sable harvested in Russia, shown for comparison. Value in US dollars, based on sales results of the [1]Fur Harvesters Auction Inc. (source: www.furharvesters.com), [2]North American Fur Auction (source: www.nafa.ca), [3]St. Petersburg International Fur Auction (http://www.sojuzpushnina.ru), January–April 2016. Numbers harvested are from the National Furbearer Harvest Statistics Database (source: www.fishwildlife.org).

Species	Value of pelt in USD Average, ('top' price)	Numbers harvested in the US, 2014*
wild mink	13.09 (15.00)[1]	57,883
otter	22.32 (39.00)[2]	20,391
raccoon	11.54 (26.00)[2]	1,426,786
wolverine	230.63 (320.00)[1]	475
ermine	2.21 (6.30)[1]	17,483
badger	15.71 (25.00)[1]	14,858
skunk	4.75 (11.00)[1]	105,352[†]
marten[‡]	50.00[2] (190.00)[2]	7883
fisher	38.54 (290.00)[2]	3123
ringtail[§]	Not listed	172[§]
Barguzinsky sable	110.83 (510.00)[3]	–
Arctic timber wolf Eastern timber wolf	262.15 (490.00)[1] 80.94 (250.00)[1]	1045 (total grey wolf harvest)
red fox	22.05 (115.00)[1]	116,982
bobcat	119.69[1] (900.00)[2]	42,766

*most recent data available; [†]includes 104,372 striped skunk, 165 hooded skunk, and 815 spotted skunk; [‡]marketed as 'North American–Canadian sable'; [§]listed as 'bassarisk', reportedly sold for a top price of USD3–4 in 2009 (www.trapperpredatorcaller.com/article-index/marketreport_february09)

American mink and striped skunks (*Mephitis mephitis*) are most frequently trapped as a secondary species in North America (AFWA 2015) but their populations appear to be resilient to harvest (Eagle and Whitman 1987) and offtakes of both have averaged approximately 100,000 through most of the 1990s and 2000s (although they are somewhat lower now, Table 7.2). Fisher and wolverine populations, in contrast, are highly susceptible to overharvest because of their low densities, extremely large home ranges, and low reproductive capacity (Macdonald et al., Chapter 1, this volume; Copeland et al., Chapter 18, this volume), and are taken in much lower numbers (Proulx 2000, Table 7.2). Powell (1979c) suggests that even low level harvests of fisher could result in localized extinction, and simulation modelling by Lofroth and Ott (2007) suggests that harvest levels of 15 of 71 wolverine population units in British Columbia between 1985 and 2004 were unsustainable. In the US, wolverines are hunted only in Alaska and Montana—in western Montana, low numbers of wolverines are found in small mountain ranges and localized trapping pressure can affect these small populations even under moderate state-wide harvest limits (e.g. Squires et al. 2007)—less than 10 individuals per year have been taken from Montana since 2006, and none in 2014 (the last year for which fur harvest data were available at the time of writing). American marten harvest is also considered unsustainable in some states (e.g. Michigan, Skalski et al. 2011) and the effects of harvest can be compounded by the impacts of forestry on habitat quality and increased accessibility to trappers via forest roads (Hodgman et al. 1994; see Cushman and Wasserman, Chapter 12, this volume). All three species are listed by the IUCN as Least Concern,

Figure 7.4 Raccoons trapped for their fur. Photo: Born Free USA/Respect for Animals.

on the basis of their wide distributions and presence in protected areas. However, at a regional level, the eastern wolverine population is Endangered, and the western of Special Concern. Conservation measures recommended in the IUCN Red List entries for fisher and wolverine are, respectively, 'to prevent excessive harvest' (Helgen and Reid 2016b) and to 'reduce legal and illegal hunting' (Abramov 2016a).

All other North American terrestrial musteloids are trapped as furbearers to a greater or lesser extent, with pelt values that range from a couple, to several hundred, US dollars (see Table 7.2; see also Gompper, Chapter 25, this volume, for discussion of the potential role of overharvest in the decline of eastern spotted skunks [*Spilogale putorius*]).

Russia and China

The Russian fur trade is dominated by sable, approximately 650,000 of which were harvested in Russia in the 2011/12 season, most sold though the international St Petersburg Fur Auction for about USD 192 per pelt (Minkov 2013; see also www.sojuzpushnina.ru). In 2016, the maximum price paid for the best Barguzinsky sable was USD 510, making it the most valuable of the musteloids in recent years (Table 7.2). A designer coat made of up to 60 Barguzin sable pelts might sell for over USD 100,000 (Tyler 2000). Sable are abundant over much of their range (Monakhov 2016), and harvest is regulated (and subject to restricted hunting periods, www.sojuzpushnina.ru), but regularly exceeds annual quotas: in 2003, an estimated 112,000 sable for auction at St Petersburg were harvested illegally, and the number of skins offered at auction has exceeded official harvest by about 50% since 1997 (Dronova and Shestakov 2005). That harvest can impact population size was evidenced by a tripling of the population within 15 years in the Middle Urals following a decline in hunting there in the early 1990s when fur trapping became unprofitable following the collapse of the Soviet Union (Korytin 2011).

Another Russian export, the Kolinsky sable, is not a sable at all but actually a Siberian weasel, the tails of which are made, not into coats, but into fine, high quality artists' paint brushes. The best brushes (used for watercolour painting and, in China, for Chinese painting and calligraphy, Bangjie 1989) are made from the hairs of the tip of the tails of male weasels in winter. Siberian weasels have a widespread distribution that includes China as well as much of Russia and most (although they are considered to be inferior to individuals trapped in Siberia) appear to be exported from China rather than Russia. However, because Siberian weasels are listed only on Appendix III of CITES, various exemptions to the regulations mean that trade between China and some countries in Europe does not require CITES permits, and thus that the 3000 to 50,000 tails that CITES record as being exported every year presumably reflects only a proportion of the total harvest. Siberian weasels are categorized as Least Concern by the IUCN but there is no population monitoring (particularly in China, Lau et al. 2010) so it is unclear if present harvest levels are sustainable in the long term (Abramov et al. 2016b).

Red pandas are strictly protected (nationally and internationally) but red panda furs (once worn traditionally throughout their range) are still commonly seen adorning the heads of Tibetans of the Yi minority in the southern Chinese province of Yunnan (Glatston 2011; Wei et al. 1999b; Ziegler et al. 2010), where newly married couples wear the hats as a talisman for a happy marriage (Wei et al. 1998, 1999b). Wearing traditional clothes was banned in a number of Chinese provinces, including the Tibet Autonomous Region, in 2008 (Ziegler et al. 2010), and it is unclear whether the hats worn are the result of contemporary hunting, or are family heirlooms. Red panda skins are still occasionally found for sale in Nepal and at trade markets between China and Myanmar (Glatston 2011; Wei et al. 1999b). Hunting for skins is considered to affect only a minority (< 50%) of the global population (Glatston et al. 2015) but, as for many of the threatened otter species (see Threats in Macdonald et al., Chapter 1, this volume), it is one of a battery of direct threats (see Hunting for food, and Pets, below) compounding the effects of widespread habitat loss and fragmentation. The smaller populations (as in, for example, Nepal, Sharma and Belant 2009; Glatston et al. 2015) can support little or no offtake.

Otherwise, there is little information on trapping wild furbearers in China. Altai (or mountain) weasels (*Mustela altaica*) are occasionally hunted and CITES trade records show export of a few hundred skins each year through the 1990s but none since 2005 – the species is declining due to habitat loss but hunting is only at low levels and is unlikely to be driving population declines (Abramov 2016c). Chinese ferret badgers (*Melogale moschata*) are reportedly subject to heavy harvest pressure in parts of China for local use (Lau et al. 2010; Duckworth et al. 2016a), but are still common and widespread, and generally thought to be resilient to trapping, as are back-striped weasels (*Mustela strigidorsa*) that are apparently not sought after but occur in areas where non-selective trapping is widely practised (Roberton et al. 2016). There seems to be little in the way of systematic monitoring for either species.

Otters—worldwide

All otter species are hunted for their skins (Table 7.1). In North America, following the widespread recovery of the North American river otter (due in part to reintroductions but also improvements in water and habitat quality, and harvest management), 37 states and all Canadian provinces except Prince Edward Island now allow commercial trapping (Bricker et al. in press). CITES trade records suggest that the vast majority of the approximately 20,000 skins harvested annually are ultimately exported to Hong Kong and China, although they are not particularly valuable (about the same as the average value of a red fox pelt, Table 7.2). All states that permit harvest report stable or increasing otter populations (Bricker et al. in press), but there is some concern that harvest could limit further population recovery in some areas (Serfass et al. 2015). CITES records very little trade in Eurasian otter skins, the only notable record being of 900 'skin pieces' exported from India to the US in 2003, but there is concern regarding poaching of this species in Asia (Roos et al. 2015), where the illegal wildlife trade poses a serious threat to all otter species (Eurasian, small-clawed [*Aonyx cinereus*], smooth-coated [*Lutrogale perspicillata*], and hairy-nosed [Lutra sumatrana], Gomez et al. 2016). Here otters are hunted illegally, and sold for relatively high prices (due presumably to their rarity, Courchamp et al. 2006). A recent report by the IOSF (2014) found that in Cambodia 'otter' skins (of unidentified species) sell for as much as USD 200 (in an area where a family might survive on a few dollars a week), and several hundred sea otters (obtained illegally from the Commander Islands Biosphere Nature Reserve) are being sold on the black market in Russia each year. In northern Myanmar, otters are now largely absent (TRAFFIC 2009), and their skins (priced at USD 102 per skin) are second only in value to a kilogram of bear gall bladder or musk deer musk (Rao et al. 2011).

Hunting for food

Musteloids are relatively rare in the bushmeat trade compared with other taxa, and even among the carnivores (that are less popular than ungulates, rodents, or primates, e.g. Fa et al. 2006), civets and genets tend to be favoured over musteloids (e.g. Doughty et al. 2014). The strong smell of musteloids (associated with chemical communication, see Buesching and Stankowich, Chapter 5, this volume) is often considered off-putting; nevertheless, over 20 musteloid species are at least occasionally hunted for food (Figure 7.3; Table 7.1).

In South China, people have traditionally relied on wild animals as a source of protein, and many small carnivores are seen for sale in food markets there (e.g. Lau et al. 2010). In the uplands of Laos, small carnivores are eaten by ethnic minorities and hunting is permitted outside core protected areas (although offtake is limited and trade prohibited, A. Johnson et al. 2009). Increasingly, across Southeast Asia and China primary demand comes from upmarket urban wild meat restaurants (Bell et al. 2004). Red panda meat is reportedly fairly widely available in such restaurants in China (Ziegler et al. 2010; Glatston 2011; Glatston et al. 2015). Elsewhere, musteloids are sometimes eaten if they are obtained incidentally but they are not specifically targeted for their meat (e.g. western Africa, Jacques et al. 2009; India, Datta et al. 2008). Local taste preferences vary: for example, hog badgers (*Arctonyx collaris*) are not eaten in northeast India (Datta et al. 2008) but are so popular in China that they are farmed there for food (Duckworth et al. 2016e; see also Lau et al. 2010), and whilst otter meat is not eaten in Siberia because it reportedly smells of raw fish (Cherkassov 2012) it is sold in the US via online exotic meat markets.

This predominantly local-level hunting and trapping for food is of little conservation concern for most of the musteloids that are eaten but is a major threat for hog badgers (Duckworth et al. 2016e), for which exploitation is considered to be unsustainable (Chen et al. 2015; Zhou et al., Chapter 13, this volume), and South American coatis (*Nasua nasua*), although the latter are otherwise widespread and apparently common (Emmons and Helgen 2016c).

Traditional medicine

Traditional healers in Africa, Asia, and Central and South America use musteloids to cure a range of ailments. Examples include the use of skunk soup in Mexico to cure anything from acne to a stomach ache, cough, or even blindness (Alonso-Castro 2014), back-striped weasels boiled in tea in Laos for backache (Hansel and Tizard 2006), African striped weasels (*Poecilogale albinucha*) in Zaire for rheumatic pain (Carpaneto and Germi 1989), dried otter penis mixed with coconut milk in Cambodia to increase sexuality

(Dong et al. 2010), and badgers in Mongolia to treat throat cancer (Soloviev et al. 2014; see Zhou et al., Chapter 13, this volume, for other uses of badgers in traditional medicine). Honey badger brains are used to treat headaches (De Luca and Mpunga 2013), their bones, skin, and hair burnt and mixed with herbs to make a body ointment used by hunters for 'fortification', and their bones put in the bathing water of a 'weak child' (Ntiamoa-Baidu 1992). Sable heart has been the main ingredient of Lixin pills (for the treatment of tachycardia, arrhythmia, and heart failure as well as rheumatoid heart disease, Li et al. 2000) in Traditional Chinese Medicine for over 100 years (Li et al. 2014).

Seventeen musteloids are listed by the IUCN as being used in traditional medicine (considered a potential threat for 10 species, Figure 7.3, Table 7.1), and probably all musteloids that occur in these parts of the world have some traditional use. However, there is no evidence that any one species is targeted for traditional purposes, or that this type of use extends beyond the local community.

Wild musteloids as pets

Somewhat inexplicably, given their preference for living in water, wild otter pups (in South America, Asia, and Africa) are increasingly being taken as pets. In Asia, for example, 800 otter pet owners are known in Jakarta alone (IOSF 2014) and 11 young otters, presumably destined for the pet trade, were recently discovered by customs officials in a suitcase at a Bangkok airport (Shepherd and Tansom 2013). Gomez et al. (2016) refer to what 'seems to be a flourishing online wild otter pet trade' based on scans of social media websites and trade fora in Vietnam, Malaysia and Indonesia.

Seven terrestrial musteloids (including such unlikely candidates as Humboldt's hog-nosed skunk [*Conepatus humboldtii*], crab-eating raccoons, and Javan ferret badgers [*Melogale orientalis*]) are considered to be potentially threatened by the exotic pet trade, and several others are at least occasionally taken as pets (Figure 7.3, Table 7.1). For example, Javan ferret badgers are sold as novelty pets in open markets in Indonesia (Kim 2012; Shepherd 2012) and there is evidence of increasing trade in red pandas as pets in China and Thailand (Glatston et al. 2015). Olingos (*Bassaricyon* spp.) are also popular, presumably (much like red pandas) because of their 'teddy bear' likeness (as the newly discovered olinguito [*B. neblina*] was described by National Geographic, Dell'Amore 2013). Legal international trade in live musteloids (as recorded by CITES) is currently dominated by kinkajous (also known as the 'honey bear') up to 100 of which are shipped annually from Guyana to the US, Japan, and other countries (Harrington 2015, Figure 7.5). Otherwise international trade in live musteloids is probably relatively infrequent (fewer than 30 individual live tayra [*Eira barbara*] and coati are traded internationally under CITES each year), and even the current offtake of kinkajous is believed to be sustainable (Brooks and Kays, Chapter 26, this volume). However, local declines are possible and any level of trade in endemic species with restricted ranges (such as the Javan ferret-badger) is a potential concern (Shepherd 2012).

Management issues

Contemporary exploitation of musteloids falls under three general scenarios: legal management as a furbearer, illegal international trade, and local use, each of which differs in the way in which it is managed or

Figure 7.5 Pet kinkajou (this individual is from captive parents but up to 100 are taken from the wild every year). © Janda Exotics.

regulated. However, in considering the potential impacts on musteloids, there are a number of common issues that are relevant.

Population and harvest monitoring

Of particular concern is the lack of population data even for relatively well-known species that are harvested legally in large numbers. For American marten, for example, although the total population size is estimated to be at least several hundred thousand individuals, adequate population data are unavailable for much of their range (Helgen and Reid 2016a). North American river otter population assessments are based on modelling rather than field data, and 29 states in which otters are harvested lack any kind of population estimate (Bricker et al. in press). Monitoring is particularly important for the larger species (such as wolverines) that suffer additive mortality and reproduce slowly (e.g. Golden et al. 2007b) but these species also tend to have low density populations in remote, rugged areas, so it is difficult to estimate population vital rates. Consequently, the impact of trapping on wolverine population dynamics is not well understood (Squires et al. 2007) and there is a risk of overharvest (Ruggiero et al. 2007). Further, even in the US, where national harvest statistics are published for pelts sold, many states do not require trappers to report the numbers of animals taken (www.bornfreeusa.org), and there are no records of unsuccessful trapping (i.e. trapper effort, Golden et al. 2007b) or poaching (Lofroth and Ott 2007). Similarly, in Russia, hunters do not always supply complete information to the authorities as to where, when, and how many sable were taken (Monakhov 2001). Even for the raccoon, a common US species that is currently increasing in population size, Zeveloff (Chapter 27, this volume) comments that harvest regulations (in contrast with those for large game animals) are made in the absence of any type of quantitative analysis.

In other parts of the world, basic information on the status and distribution of many of the small carnivores is lacking and so it is impossible to know if harvests of hunted species are sustainable (e.g. in the case of the Siberian weasel). For the apparently rare species, one logical, conservative approach to dealing with this lack of data, introduced in the regulations for a newly protected area on the Laos–Vietnam border, is the prohibition of hunting of ‘rarely detected’ (as well as globally threatened) small carnivores (A. Johnson et al. 2009).

There are few quantitative data on the use of animals for traditional medicine or other cultural uses, and these uses are very difficult to monitor, since body parts usually come from unknown locations and represent an unknown number of animals (Whiting et al. 2011). However, monitoring these local-level uses is probably of relatively low priority for most species.

Refugia

The creation of refuge areas (untrapped wilderness areas) is essential for many species (e.g. wolverine, Weaver et al. 1996; Magoun and Copeland 1998; Proulx 2000; Krebs et al. 2004; Golden et al. 2007b; American marten, Wiebe et al. 2013; and sable, Monakhov 2001) to protect breeding individuals, and to provide immigrants for local sink areas depleted through harvest (Hodgman et al. 1994). The effectiveness of such areas will be dependent on their size (Squires et al. 2007), accessibility to trappers (Wiebe et al. 2013), and future habitat encroachment by humans (Golden et al. 2007b). In Russia, remote, difficult to access areas, that are no longer profitable to trap, serve the purpose of refuge areas for breeding populations of furbearers that are heavily exploited elsewhere (Dronova and Shestakov 2005).

Trapper motivation

Trapper motivation varies (regionally and according to exploitation type) and has implications for how trapping is regulated, and how it may change over time. In the Russian Far East, for example, trapping for furs remains a strong family tradition and plays an important role in supporting local livelihoods (Dronova and Shestakov 2005): for 25% of 308 hunters interviewed, hunting was their main income-generating occupation (although 50% of hunters mentioned ‘passion for hunting’ as well as financial interests). Most hunters in this area expect to harvest 50–70 sable per season. In contrast, in North America, most trappers trap for recreation and the ‘outdoor experience’ (although pelts are still sold), but state-managed recreational trapping is widely considered a core element of wildlife conservation management (see Organ et al. 2012). Some conservationists question whether the focus on hunting is the right approach for conservation in the twenty-first century (e.g. Nelson et al. 2011; Peterson and Nelson 2017; T. Serfass, pers. comm.) and Organ and Fritzell (2000) suggest that declining numbers of hunters in the US through the twentieth century may lead to hunting policies becoming more aligned with broader societal mandates.

When otters that come into conflict with local fisheries (discussed in the second part of this chapter) are killed, selling their skins and/or body parts is sometimes viewed as partial compensation for damage suffered even though the amounts recouped may be minimal. Spotted-necked otter (*Hydrictis maculicollis*) bodies (with heads) fetch up to USD 33 in Benin, and a foot (sold at fetish markets) fetches a mere USD 2–5 (2008 prices) (Djagoun and Gaubert 2009)—relatively poor compensation for damage caused to fishing equipment that may amount to over USD 200 (Akpona et al. 2015).

Young musteloids taken as pets are sometimes taken opportunistically when hunting adults for their pelts (e.g. Schenck and Staib 1998) and it is unlikely that local use as pets is a significant motivator for trapping. However, popularity of these small carnivores as unusual pets (e.g. in the US) and a potentially lucrative and increasing international pet trade (see e.g. Bush et al. 2014; Moorhouse et al. 2016) might drive higher trapping levels in future.

Cross-border trade and legal enforcement

In Asia, illegal cross-border trade, for example between Nepal, India, and China, and between Myanmar National Parks and China, is a significant threat (Shepherd and Nijman 2007; Rao et al. 2011), particularly for Asian otters, but also red pandas, and there is an urgent need to address how best to tackle it (Lau et al. 2010). Existing legislation is not always enforced: the IOSF (2014) identified 39 cases that involved otters (mostly skins, and mostly of unidentified species) since 2000, involving over 8000 individuals—only 12 cases culminated in arrest. Local research, sound scientific data, education, and better public awareness are all needed, but only with international collaboration and transboundary cooperation will real progress be made.

Whether for fur, food, or pets, animal welfare concerns are paramount, and we return to this issue at the end of the chapter.

Conflict between people and musteloids

As small, or meso-, predators, musteloids have all the capabilities of the 'big carnivores' (albeit in smaller packages) but are also small and adaptable enough to live among humans in agricultural environments, sometimes even in houses in towns. Thus, musteloids can also cause problems for people—both real and perceived.

Human–wildlife conflict is typically characterized as arising when the behaviour of wildlife poses a direct threat to the livelihood or safety of a person or a community that may result in persecution of the species (Inskipa and Zimmerman 2009). In that sense, musteloids differ somewhat from the larger carnivores in that, by virtue of their smaller body size, they only rarely pose a direct threat to humans, and only wolverines ordinarily kill livestock. Predation on small domesticated, stocked, and game animals (terrestrial and aquatic) is instead the most commonly reported cause of conflict between musteloids and people (Figure 7.6a). Otherwise, musteloids in conflict with humans fall into a kind of 'agricultural or urban pest' category where they can cause damage to crops, buildings, and even cars (Figure 7.6a). Sometimes musteloids also cause problems by preying on other endangered or threatened species—which, although not usually considered under the umbrella of 'human–wildlife conflict' does create a conflict of interest insofar as these small predators threaten other wildlife that some people want to protect.

Musteloids are thus perceived as thieves of personal property, a cost to farmers and gamekeepers, competitors for a shared resource, a nuisance, and (under some circumstances), a threat to biodiversity, with the species most commonly reported in conflict with people being martens, otters, raccoons, and badgers (Figure 7.6b). In this section of the chapter, we give an overview, and provide examples, of some of the different types of conflict, giving an estimate of the extent, cost, or seriousness of the damage sustained wherever possible, before going on to consider the consequences for musteloids, and potential non-lethal solutions.

Predation

Domestic poultry, gamebirds, and nesting turtles

Musteloids everywhere (including the lesser-known species, such as lesser grison, Brooks 1991; striped polecats [*Ictonyx striatus*], Stuart et al. 2015a; and Patagonian weasels [*Lyncodon patagonicus*], Kelt et al. 2016) eat poultry. Chicks kept in small runs are especially vulnerable to weasels that can enter supposedly predator-proof enclosures through mole tunnels (King and Powell 2007). Raccoons rob eggs from chicken coops in gardens (Kays and Parsons 2014), and stone martens (*Martes foina*) seek eggs in villages and attack chickens that are not locked up at night (Herr 2008). Honey badgers are so strong and dexterous that they are difficult to keep out of poultry pens (Rosevear 1974). Of 66 gamekeepers surveyed in England, 70% of those rearing game birds in pens experienced polecat predation, and over 70% of those reported surplus killing of over 20 birds in a single

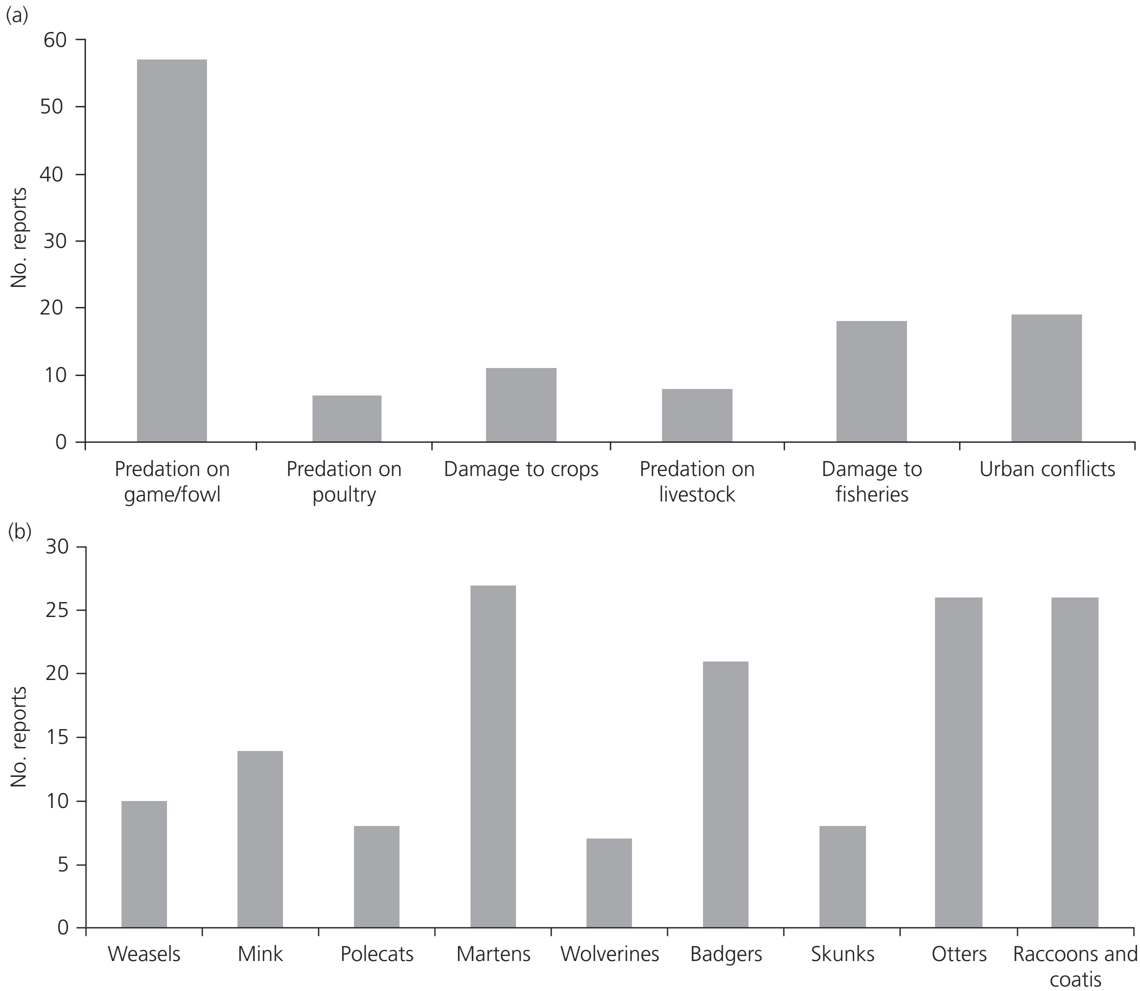

Figure 7.6 Number of publications with reports of a) various types of conflict between musteloids and people, and b) species groups involved in the conflict. N = 264 scientific papers (sourced from Web of Knowledge), 1970–2012.

night (at one site, 120 birds were taken in one night, Packer and Birks 1999). There are few data on the actual costs of losses, however, and losses are often minor compared with those due to, for example, foxes (e.g. Stahl et al. 2002).

In the US, raccoons and striped skunks eat the eggs of various sought-after wild game birds (e.g. Jennings et al. 2006; Larivière and Messier 1998a, b, 2001b), mink kill waterfowl ducklings (Amundson and Arnold 2011; Zoellick et al. 2004), and American badgers (*Taxidea taxus*) destroy sage-grouse nests (Coates and Delehanty 2010). In Scotland, pine martens (protected in the UK) cause controversy by eating locally vulnerable capercaillie (*Tetrao urogallus*, a large ground-nesting game bird) eggs. Capercaillie are also impacted by fox and crow predation, and by climate, and the relative importance of each is not clear; nevertheless, losses due to pine martens can be high (up to 57% of clutches), particularly in areas where red foxes and crows are controlled (Summers et al. 2009).

Predation impacts can be magnified if mesopredator density becomes atypically high in the absence of larger predators ('mesopredator release', Crooks and Soulé 1999; and this appears to be the case for raccoons and skunks in some regions of the US), in urban areas where humans provide supplemental food sources (Jokimäki and Huhta 2000; Jiménez and Conover 2001), or where habitat and landscape modifications reduce cover for prey and facilitate access for predators. Irrigation in the western US, for example, has facilitated the spread of medium-sized predators into areas where they did not previously exist, because it was too dry, and the subsequent creation of wetland refuges for waterfowl has become a haven for the mesopredators that prey upon these species, their young, and their eggs (Frey and Conover 2006; see also Krapu et al. 2004). On sandy beaches along the US coast, sea turtle nesting habitat has been greatly reduced by coastal development but raccoon populations thrive in proximity to humans and reach densities as high as 238 raccoons per km^2 (Engeman and Smith 2007). Raccoons and skunks can locate buried turtle eggs within a few days of nest construction (Vilardell et al. 2012) and, although it is not entirely clear how they find them (Burke et al. 2005), up to 95% of nests may be depredated (Engeman et al. 2012; see also Wirsing et al. 2012).

Fisheries

In the pond fisheries of Central and Eastern Europe (traditionally dominated by cyprinid culture, mainly common carp [*Cyprinus carpio*]), fish production has suffered from increasing levels of otter predation and disturbance over the last couple of decades (e.g. Adámek et al. 2003; Kloskowski 2005, 2011). In a study in southeast Poland, 94% of 110 farms were frequented by otters, and, although other piscivores also took fish, only otters were reported to take the large 'marketable' fish (Kloskowski 2011). In this part of Europe, otter diet can be comprised almost entirely of carp (e.g. Kortan et al. 2007; Baltrünaité 2009), and, in the Czech Republic, losses caused by otters on fish farms (in 1999) were estimated to total almost a quarter of a million dollars (Adámek et al. 2003). On the Californian coast in the US, recovering sea otter populations were associated with a tenfold decline in red abalone density and the subsequent collapse of a previously valuable commercial fishery in the 1960s (Wendell 1994), and, where sea otters occur now, red abalone occur at lower densities (0.1–3.6 individuals per 20 m^2 plot where sea otters are present compared with 11.5–18.1 where they are not, n = 4 sites each), are dominated by smaller individuals (58–99 mm shell lengths vs. 142–190 mm) and are largely restricted to crevices where they are protected from predation (Fanshawe et al. 2003). All commercial abalone fisheries are now closed, and their restoration is considered incompatible with sea otter restoration (with serious implications for marine protected area planning, Fanshawe et al. 2003; Chadès et al. 2012). There are often similar complaints of otter interference in subsistence fisheries in parts of South America and Africa; certainly, otters can cause costly damage to fishing gear (e.g. Akpona et al. 2015) but, in natural river systems, overlap between the fish species preferred by otters and those taken by people is, in reality, often rather low (e.g. Butler and Marshall 1996; Rosas-Ribeiro et al. 2011; see also Recharte et al. 2008). In the UK, where recreational or sport fishing is a popular pastime, fishermen consider otters a nuisance and are affronted to find partially eaten carcasses of large specimen fish (Figure 7.7) that might be up to 60 years old, known by name, and potentially worth several thousand dollars. These large carp appear

(a) (b)

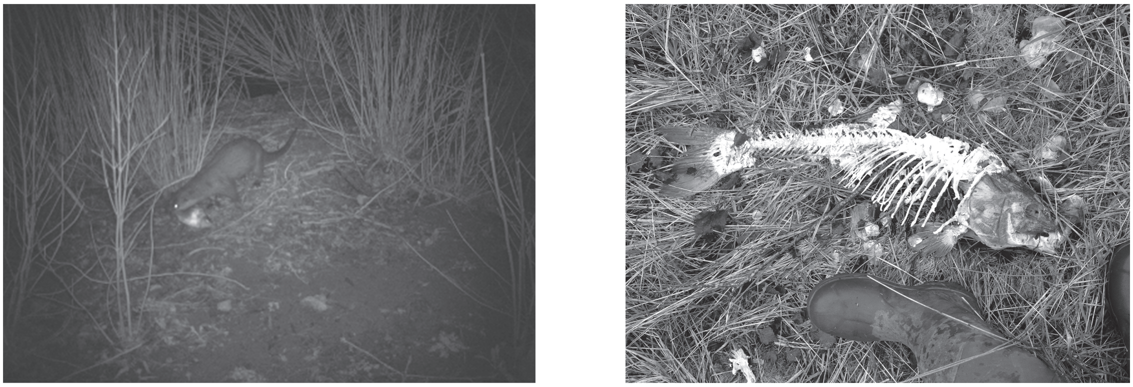

Figure 7.7 a) Camera trap photo of an otter with a fish on a small lake island, b) remains of a carp carcass on the lake bank, both at private fishing lakes in southern England. © A.L. Harrington, L. Harrington, respectively.

only occasionally in otter diet, and are probably difficult to catch most of the year, but are vulnerable to predation during cold winter periods when they enter torpor and rest motionless on the lake bottom (Britton et al. 2005; Grant and Harrington 2015).

Livestock

In Ethiopia, honey badgers occasionally kill goats, sheep, and chickens kept by villagers but the extent of the problem is small compared with that due to local crop raiders (predominantly monkeys, Mojo et al. 2014). European badgers are also occasionally reported to attack and kill lambs and poultry in England and Wales but incidents are not common and rarely attributable to badgers with any certainty (Moore et al. 1999). Wolverines kill livestock but only in Fennoscandia. Here, indigenous Sámi pastoralists herd and harvest free-ranging semi-domesticated reindeer (*Rangifer tarandus*) in the north, and each summer 2.5 million unattended sheep roam over tens of hundreds of square kilometres of Norwegian uplands. Between 2000 and 2012, wolverines were responsible for the loss of almost 2000 reindeer (about a third of all documented losses) in Norway (Tveraa et al. 2014) and, during the summer grazing season, they can each kill two to five lambs in a single day (Warren et al. 2001; Landa et al. 1999). It is estimated that almost 10,000 sheep are killed by wolverines in Norway each summer (Abramov 2016a).

Agricultural damage

Only common raccoons, honey badgers, and European badgers are implicated in agricultural damage of any real significance. In the US, for example, raccoons can each destroy 8–12 corn (*Zea mays*) husks per night (Stern et al. 1996) and the damage caused to commercial corn crops may even exceed that caused by deer browsing (DeVault et al. 2007; Beasley and Rhodes 2008). Raccoon-caused crop damage has increased in recent decades, particularly in the midwestern US, presumably related to increases in raccoon abundance in this area (see also Macdonald et al., Chapter 1, this volume), and costs to farmers are predicted to be substantial where fields border forest patches that support high densities of raccoons (Beasley and Rhodes 2008).

In South Africa, honey badgers feed on bee larvae and honey and damage domestic beehives. Beekeeping is an important component of South Africa's agricultural industry, bringing in an estimated 16 million dollars' worth of honey and other bee products (Begg and Begg 2002), and providing pollination services estimated to be worth over 500 million dollars (Allsopp 2000), per annum. Although reports of the honey badgers' predilection for honey are largely mythical, domestic hives are (for a honey badger) an easy and readily available source of food, and individuals can destroy more than 20 hives in a night (Guy 1972). The estimated cost of damage for just one hive (in damage, loss of pollination services, loss of production, and replacement costs) is estimated at USD 115 (Begg 2001). In the late 1990s in England and Wales, European badgers were estimated to cause the equivalent of 30–60 million dollars' worth of damage to cereal and vine crops each year (due to both eating and trampling them), in addition to the structural damage to fences and buildings caused by their burrowing activities (Moore et al. 1999). In some parts of the UK, the Department for Environment, Food, and Rural Affairs (Defra) receive more complaints regarding badgers than any other mammal or bird (Poole et al. 2002) but only 5% (of almost 2000 survey respondents) incurred costs of more than USD 1600 (Moore et al. 1999).

At smaller scales, raccoons in North American damage corn crops in gardens (Conover 1987); badgers raid carrots from garden allotments in the UK (Harris 1984); South American coatis, crab-eating raccoons, and tayra eat fruits from plantations in South and Central America (Aguiar et al. 2011; Soley 2012); and raccoon and striped skunk eat avocados (Borchert et al. 2008). But these incidents are little more than a 'nuisance', if that. Orchard avocados in southern California, for example, are consumed by wildlife only as fruitfall, and the estimated annual losses never exceed 2% (Borchert et al. 2008).

Urban pests

Musteloids in urban areas are, for the most part, annoying to residents (rather than costly or dangerous). European badgers, raccoons, and striped skunks in search of food in residential areas create a mess by 'dustbin rifling' (Harris 1984; Gehrt 2004); skunks (and badgers) dig up lawns in search of insects and earthworms (Rosatte et al. 2010), raccoons are noisy, and skunks are accused of being unpleasantly smelly (Gehrt 2004). European badgers occasionally kill pet guinea pigs kept in gardens (Harris 1984) and raccoons en masse sometimes kill cats (Associated Press 2006), but these incidents are rare compared with, for example, coyotes (e.g. Foderaro 2015; Poessel et al. 2013; Alexander and Quinn 2011).

Structural damage caused by den building is more serious and certainly more costly. For badgers, which have a propensity to dig setts in or near gardens and under man-made structures such as sheds, patios, and roads, the estimated cost of damage *per case* in England in the early 2000s was USD 1613 (and as much as USD 968,000 to exclude badgers from roads and repair subsidence damage)—Natural England (the government licensing body) during the same period received *c.*700 applications per year from householders or other urban landowners for licences to interfere with badgers or their setts for purposes of damage prevention (Central Science Laboratory 2007). Stone martens and raccoons (Figure 7.8) construct dens in roofs (Herr et al. 2010; Riley 1989) and, in Bern, Switzerland, there were 635 insurance claims for stone marten damage to roof insulation over four years in the early 2000s, costing almost 200,000 dollars per year, with an estimated country-wide cost of 1.6 million dollars (Kistler et al. 2013). In Scotland, 44.3% of natal pine marten dens in rural locations were in buildings such as barns, sheds, and outhouses (Birks et al. 2005).

The tendency for stone martens in central Europe to get inside car engines and to bite rubber cables and tubes is as puzzling as it is costly. Possible explanations include thermal benefits, a safe refuge or resting site, or cub play behaviour. Herr et al. (2009b) suggests that car 'visits' are simply part of the martens' normal patrolling and scent-marking behaviour (and perhaps heightened territoriality in spring—when most damage occurs, Lachat 1991); Kugelschafter (cited in Herr et al. 2009b) suggests stone martens have an affinity for volatiles released from warm rubber or plastic components. There has been no further support for Kugelschafter's suggestion but an intriguing news article reported a wolverine that had chewed the rubber parts of two chainsaws it had taken from a forestry site in Finland (Yle News 2013). In Germany in 1998, 160,000 cars were damaged by stone martens, at an estimated total cost of over 20 million dollars (Langwieder and Höpfl 2000, cited in Herr et al. 2009b).

Consequences of conflict

The consequences, for musteloids, of this array of potential conflict scenarios depends on the response of people to the situation, which is influenced by their perceptions of the severity and extent of the problem, and by their attitudes towards the species, which, in turn, varies among stakeholders and species involved. Here we describe, as an example, the variable attitudes towards three of the larger musteloids in the UK (including two that had virtually disappeared from England and that have now recolonized much of their former range—polecats and otters). We then summarize, using the IUCN Red List, the musteloid species that are persecuted as a result of conflict and outline some of the issues associated with the lethal management of non-threatened musteloids.

Perceptions, attitudes, and persecution

In England and Wales, over half of farmers who experienced agricultural damage by badgers said they

Figure 7.8 Raccoon removing cedar shingles from a roof. Common (or northern) raccoons are said to cause *c.*60% of all urban wildlife damage in the US (de Almeida 1987). © S. Miller

were tolerant of them (Moore et al. 1999) and in many badger damage cases in Britain damage is not severe enough that, for example, installation of electric fences would be cost-effective (Roper et al. 1995). Similarly, almost 70% of gamekeepers in the 1990s regarded the polecat as a 'minor pest' and a lesser threat than foxes or feral cats (although stoats also ranked higher) (Packer and Birks 1999). However, most (91%) trapped up to 40 polecats each year. Farmers that did not rear game birds were even less concerned about polecats, did not view them as a predator of economic significance, and believed that they helped control rabbits and rodent pests on the farm. Nevertheless, both farmers and gamekeepers expressed concern regarding the increasing number of polecats and wished to be allowed to control them.

Fishermen and stillwater fishery owners in England appear to be less tolerant towards otters, although perhaps only due to the views of an outspoken few (e.g. McFadden 2014). In Scotland, where otters did not decline and have always been present, fishermen appear to be more tolerant (K. Baird et al., unpublished data). In England and Scotland, it is unlikely that conflict leads to otters being killed by fish farmers and there are strict laws against it, although a few are probably trapped or shot (Chadwick 2007). In southeast Poland, private pond owners perceived losses to be higher (given similar levels of otter presence) than did managers of state-owned fisheries, and 17% of 65 fish farmers that reported significant otter predation of fish stocks admitted that otters were illegally killed on their farms (Kloskowski 2005). In other European countries, despite strict protection, otters are occasionally shot (Elmeros et al. 2012), and elsewhere in the world, otter persecution is a serious problem (e.g. for giant otters, Rosas-Ribeiro et al. 2011; dos Santos Lima et al. 2014). At least nine of the 13 otter species are killed as a perceived threat to local fisheries.

In mainland Europe, stone martens are sometimes persecuted as pests (Abramov et al. 2016a) and in Texas, shooting of skunks around houses is the single biggest cause of mortality for the species (Hansen et al. 2004) (although, where it occurs, it is not considered a threat at the population level). In rural areas, American badgers are trapped, shot and poisoned because the burrows they dig are considered a hazard to livestock and vehicles (Helgen and Reid 2016e). Many species are at least occasionally killed for eating poultry.

According to the categorization of species' threats in the IUCN Red List (Version 2016.3), at least a quarter of all musteloids are potentially threatened by persecution and/or control. For the terrestrial species, most of those threatened by persecution or control are not 'threatened' species per se (Table 7.1) and the extent of persecution generally is considered to be of low or negligible impact. There are some exceptions. Honey badgers, for example, are persecuted by apiculturists throughout their range; they are listed as Least Concern but are vulnerable to population decline due to their low reproductive rate (Begg et al. 2005a). In Sweden, poaching (to reduce reindeer predation) is the single most important cause of mortality for adult wolverines that are listed as Vulnerable in Europe (Persson et al. 2009). Otherwise, lethal control of non-threatened musteloids is used as a legitimate method for reducing predation in game management and in sea turtle conservation, and for reducing the abundance of urban musteloids where they are causing problems. In Illinois, for example, over 10,000 raccoons were killed every year in the 1990s (Bluett et al. 2003, see also Hadidian et al. 2010) and numbers are presumably similar in other US states.

Lethal management of non-threatened musteloids

Shooting gamebirds is still a major and economically valuable participant sport in both the UK and the US; the modern face of game preservation is, however, very different from that of its nineteenth-century forebears and indiscriminate year-round lethal predator control is now very rare. In the UK, for example, the number of full-time keepers was reduced to fewer than 5000 by the 1980s (compared with 25,000 at the turn of the century, Tapper et al. 1982) and one popular and cost-effective policy now is simply to rear large numbers of birds artificially and release them shortly before the shooting season. Hence the few gamekeepers that are left spend most of their time on rearing chicks, and tending just a few traps around the pens while the birds are young (McDonald and Murphy 2000).

In a review of 83 predator removal studies (covering 128 prey species from six continents), R. K. Smith et al. (2010) concluded that removing predators increased hatching and fledging success (with average increases of 77 and 79% respectively) and breeding population size of birds (71% increase). Predator control can also have quantifiable benefits for nesting marine turtles: shooting raccoons and armadillos at night reduced nest predation from 95% to less than 15%, and resulted in an additional 128,000 or more hatchlings emerging (at an estimated cost of only USD 0.09 per additional hatchling, Engeman et al. 2012). More of the predator population must be removed than can be replaced, at least during prey nesting seasons (King and Powell 2007). If successful, this has only a local and temporary effect on the more prolific musteloids (stoats, weasels, raccoons) but does make a difference for the prey. In some cases, there can also be spillover benefits—trapping raccoons to reduce predation of pheasants (an upland gamebird) in Wyoming had the additional benefit of reducing corn damage in neighbouring fields with a revenue increase of USD 10.75 per ha (Anderson et al. 2013).

However, predator removal (for a number of reasons) is not always effective, even when removal rates are sufficiently high (e.g. Ratnaswamy et al. 1997). In Florida, loggerhead sea turtle (*Caretta caretta*) egg predation at a beach where raccoons had been controlled for 25 years was almost twice as high as at two beaches where raccoons had only been trapped in the previous year, and 30 times higher than on a beach where raccoons had not been removed but nests were protected with cages (Barton and Roth 2008). Where raccoon density was lowest, the density of ghost crabs (*Ocypode quadrata*, another common egg predator) was highest and crabs were larger. Stable isotopes indicated that the largest crabs were feeding at higher trophic levels (and probably themselves consuming more turtle eggs, Barton and Roth 2008). Similarly, in North Dakota increasing the proportion of successful mallard nests by seasonal non-specific nest predator removal resulted in negligible increases in recruitment (Amundson et al. 2011, 2013), and each incremental mallard came at a cost of USD 74. In this study, nest success (even in the presence of predators) was high and population size was instead limited by duckling survival. Ducklings were killed by American mink and raptors, neither of which were effectively targeted by generalized predator control. Identification of the most influential predator and an understanding of the role that they play in the wider ecosystem is crucial.

Positive effects, when they are achieved, can be short-lived due to influx of animals from surrounding areas ('compensatory immigration', sensu Turgeon and Kramer 2012) and other compensatory mechanisms. Long term removal may simply remove a surplus dispersing from source populations (e.g. Barton and Roth 2007) and continual management is required to remove new predators that move in. In Norway, for example, where about 14% of the country's wolverine population is harvested every year according to annual government harvest quotas in an effort to reduce livestock losses (in contrast to Sweden where they are illegally poached, Persson et al. 2015) rapid recolonization (from Sweden, Gervasi et al. 2015) negates long-term effects on lamb losses (Landa et al. 1999; Gervasi et al. 2015). (In addition, this level of harvest is believed to be unsustainable, Abramov 2016a; Sæther et al. 2005; Lambin et al., Chapter 4, this volume.) Connor and Morris (2015) liken control efforts within discrete areas to spatially structured harvest (cf. McCullough 1996), whereby harvest and refuge areas exist across the landscape. In this context, culling is particularly problematic in fragmented urban (bird or turtle) nesting areas (see e.g. Meckstroth and Miles 2005) where high densities of predators are sustained nearby. For disease-carrying animals (see Newman and Byrne, Chapter 9, this volume), there can be further negative effects of predator control insofar as culling may increase the spread of disease due to increased movements of animals ('perturbation effect', Carter et al. 2007) and this is a particular issue for the control of badgers in the UK (see Woodroffe and Donnelly, Chapter 20, this volume).

On ethical grounds, lethal control remains controversial (R. K. Smith et al. 2010), and it is often viewed negatively by the public (Goodrich and Buskirk 1995). Culling protected predators is particularly controversial—a six-year study in Spain demonstrated a doubling of productivity in the Spanish subspecies of the capercaillie (*T. u. aquitanicus*) in response to marten removal (Moreno-Opo et al. 2015) but, in the UK, whilst gamekeepers demand a pine marten cull (to reduce capercaillie predation), conservationists argue that it is not acceptable (e.g. Miller 2012; Anon 2014).

Non-lethal conflict management

Given the extent of potential conflict between people and musteloids (described above) and the problems related to the effectiveness and sustainability of lethal control (as well as its—sometimes—controversial nature), there are surprisingly few reports of alternative non-lethal solutions in the literature. In a systematic review of the literature (see Figure 7.6) carried out by one of us (J. Marino), only 73 of 264 papers on conflict between musteloids and people, published between 1970 and 2012 referred to non-lethal management methods. Non-lethal methods of reducing problems caused by 'problematic wild animals' can be broadly divided into four general approaches: moving them (translocation), scaring them away (deterrents), excluding them, or encouraging them to go (feed) elsewhere ('diversionary feeding'). Otherwise, conflict can potentially be reduced by dealing with the people affected rather than the problem per se, by paying compensation for damage suffered (reviewed in Dickman et al. 2011; although, for the musteloids, this approach seems to be used only for wolverine and otter predation). Insofar as the scientific literature reflects usage or research interest in these different approaches for musteloids, Marino's review revealed that deterrents and other aversive methods were reported most often among possible non-lethal solutions (21 of 73 papers), followed by fencing or other exclusion methods (17 papers).

Translocation

Translocation of musteloids (for management purposes) is reported relatively rarely in the literature five of 73 papers, but it is a common approach in urban and suburban areas where lethal control is logistically difficult and/or socially unacceptable (e.g. Craven et al. 1998; Mosillo et al. 1999). In the US, hundreds of thousands of raccoons are moved around the landscape nationwide by government Wildlife Services, private Nuisance Wildlife Control Operators, and the public each year (Craven et al. 1998). However, although there are various (at least partial) sources of data available on numbers translocated (e.g. www.aphis.gov.uk), there appears to be little information available on where they are translocated to, little (if any) monitoring of their fate, or whether or not, and for how long, the problem was resolved (Massei et al. 2010; Hadidian et al. 2010).

The problem is that translocated individuals often return and become problems again (Curtis and Hadidian 2010; Herr et al. 2008), or (like with lethal control) are quickly replaced by other individuals (e.g. Herr 2008), and may cause 'new problems' where they are released (Rosatte and MacInnes 1989; Craven et al. 1998; Mosillo et al. 1999). Urban raccoons and European badgers, for example, prefer familiar habitat when translocated and seek gardens and houses for foraging and denning post-translocation (Brown and Cheeseman 1996; Mosillo et al. 1999; Herr et al. 2008), and translocated female raccoons use chimneys for birthing and nursing dens even when there are natural tree cavities available nearby (O'Donnell and DeNicola 2006).

Non-lethal methods offer a seemingly humane option and are often preferred by the public, but there are still welfare issues. Trapping is stressful (and can result in physical injury, Craven et al. 1998), as is being moved to an unfamiliar area where animals may not be able to find a new territory in a novel (and potentially already occupied) area. Post-translocation survival may be low (e.g. raccoons, Rosatte and MacInnes 1989; stone martens, Herr et al. 2008; but see Mosillo et al. [1999] who found that translocated raccoons survived well, and Brown and Birks [2006] who describe successful removal of pine martens from houses). And there is usually little consideration of the impacts of translocation on conspecifics or other animal populations at the release site (Craven et al. 1998). For raccoons, that are abundant across much of their range, there are few areas suitable for translocations that are not already occupied by other raccoons, and the risk of spreading disease is a serious concern (Mosillo et al. 1999).

Deterrents

Loud noises, flashing lights, and repugnant odours are all potential methods for 'scaring' animals away, most often in an attempt to deter animals from gardens, crops, and houses. Odours include moth balls in attics (Herr 2008) and rags soaked in dog urine (Brown and Birks 2006), and various commercially produced scents can be purchased online (for example, 'eviction fluids' designed to deter raccoons from chimneys, Vaantassel et al. 2013, and 'anti-marten' spray). Motion-activated sprinklers can deter skunks from gardens (Rosatte et al. 2010), and water jets decreased bait consumption by badgers at experimental plots (but only by 12% compared with control plots, Ward et al. 2008).

Ultrasonic deterrents are commonly used in the marine environment to reduce depredation of fish in nets by seals and to reduce bycatch of marine mammals (Schakner and Blumstein 2013). They have been widely marketed as bird-scarers for many years (Bomford and O'Brien 1990), but are less commonly used for musteloids. One exception (which was unsuccessful) was a test of ultrasonic deterrents to keep sea otters away from important shellfish harvesting areas (Wendell et al. 1996).

Across all taxa, the effectiveness of acoustic devices appears variable, and is dependent on the device, the specific sound used, and acoustic output in the field, as well as a range of external factors, such as food and motivation (A. Harrington et al. 2013). There tend to be few experimental tests of the effectiveness of deterrent devices and commercially available deterrents (scent or sound-based) are often ineffective. For example, Kugelschafter et al. (1997, cited in Herr et al. 2009b) found captive stone martens to be indifferent to any kind of strong-smelling or bitter-tasting substances (including 'anti-marten spray'), and martens were not deterred from cars by ultrasound devices under the car bonnet (Ludwig 1999, cited in Herr et al. 2009b).

'Success', when achieved (for any deterrent), can be short-lived due to habituation, and the presence or absence of an alternate food source is key. Wolverines initially avoided lambs treated with volatile repellents (Landa and Tømmerås 1996, 1997), but when all lambs in a flock were treated wolverines still ate them (Landa et al. 1998b). Worse, deterrents can have a 'dinner bell' effect: Ward et al. (2008) found that ultrasonic devices *increased* badger feeding at experimental plots.

A commercially available 'electric shock' system that can be installed inside car engine compartments offers some promise for keeping stone martens out of cars, but is illegal in some countries (Herr 2008). In urban environments, a simple solution to unwelcome raccoon visits is to keep a dog in the garden (Kays and Parsons 2014), and experiments investigating the effect of fear of larger predators on raccoons (Suraci et al. 2016), in which researchers demonstrated persistent reduced foraging by raccoons exposed to 10 s predator playbacks, and subsequent increases in their prey, suggests that large carnivore vocalizations could potentially be used as a deterrent. The aim of Suraci et al.'s work was not practical human–wildlife conflict solutions, but it nevertheless has ramifications worthy of investigation in this context.

A more complex approach involves 'training' an animal to avoid a particular food type by providing it with similar food treated with an undetectable substance that causes nausea or illness, such that the animal later associates the taste of the untreated 'food' (which may be live prey, eggs, or crops) with illness, and avoids it ('conditioned taste aversion' [CTA], reviewed in Baker and Macdonald 2015). Raccoons avoided live chickens following exposure to chicken carcasses treated with lithium chloride (Nicolaus et al. 1982) and raccoons and striped skunks were deterred from eating (untreated) chicken eggs by emetine-treated eggs distributed in the area prior to the onset of laying (but not once laying had begun, Conover 1990; see also Semel and Nicolaus 1992). Baker et al. (2008) demonstrated that the addition of an external cue (e.g. an odour) can discourage animals from sampling (tasting), and thus damaging untreated 'food': badgers previously exposed to maize cobs treated with ziram (a fungicide used as a repellent against birds, and also an irritant) and a clove oil odour later avoided untreated maize cobs in the presence of the same odour. There is clearly potential for the use of CTA in wildlife management, but the logistical difficulties mean that practical application has thus far been limited.

Exclusion

A simpler solution may be to remove the problem. Martens and urban raccoons, for example, can be excluded from buildings by blocking access points and using chimney caps (Curtis and Hadidian 2010; O'Donnell and DeNicola 2006; Herr 2008; Brown and Birks 2006). Fences (electric or non-electric) have been used to protect crops (Conover 1987; Poole et al. 2002), gardens (Rosatte et al. 2010), buildings (Kistler et al. 2013), and fish ponds (Santos-Reis et al. 2013) with some success. The likelihood of animals crossing the fence is reduced with higher voltage but increased when motivation to cross is high (Poole et al. 2004; attempts to exclude badgers from their setts, for example, are often unsuccessful, presumably because they are determined to get back in, Ward et al. 2016).

Fences around waterfowl (e.g. LaGrange et al. 1995 and references therein) and turtle (e.g. L. L. Smith et al. 2013; Buzuleciu et al. 2015) nesting areas (or predator exclusion cages around nests) can reduce nest loss due to raccoon and skunk predation, and increase survival of hatchling and juvenile turtles. In a comparison of lethal control, CTA, and nest cages, only nest cages were effective in reducing sea turtle nest depredation by raccoons (Ratnaswamy et al. 1997, although raccoons can dig under the cages and there is a risk that raccoons use the cages to locate nests, Mroziak et al. 2000). A systematic review of 16 nest predator exclusion studies by Smith et al. (2011) revealed that on average the presence of fences or nest cages resulted in a 94% increase in hatching success of birds (a larger pooled effect size than for predator removal).

There are difficulties—weasels and stoats are difficult to exclude because they are small enough to get through anything but the smallest mesh fences (LaGrange et al. 1995), martens can climb over fences, and are otherwise quick to learn how to get through (Vilardell et al. 2012), and badgers can dig underneath. Nevertheless, these are design issues, and failure is often due to deficient design, inadequate maintenance (Wilson 1993), damage (Connolly et al. 2009), or poor husbandry (e.g. access holes left open at night, Packer and Birks 1999). Advice and design specifications are readily available for a range of species in, for example, the UK and the US (see also Honda et al. 2009; and there are surely lessons to be learnt from New Zealand, where fences are used to maintain mainland 'islands' free of invasive stoats, see King et al., Chapter 10, this volume). Fencing can be expensive (in which case, the problem comes down to: who pays for it?) and may not be practical for protection of large areas (e.g. extensive fish farms, Kloskowski 2005). Simple inexpensive solutions can, however, be effective in some situations: for example, beehives can be protected from honey badgers by raising the hives 1 to 1.5 m on stands made of discarded tyres or steel drums (Begg and Begg 2002). 70% of 50 commercial beekeepers interviewed by Begg and Begg (2002) in the Western Cape Province, in South Africa, reported no further damage once such techniques were put in place, and only 6% of those who had begun using these techniques continued to trap honey badgers.

Diversionary feeding

An alternative method of reducing predation rates is to provide the predator with supplemental food and thus attract them away from protected or domestic prey and reduce the need for them to seek food elsewhere. Predation on waterfowl nests by striped skunks was lower (34% vs 73%), and nest success higher (50% vs 23%), when skunks were provisioned with carp and mink chow but only when unpredictable natural food sources otherwise forced skunks to forage widely (when they encountered nests; Crabtree and Wolfe 1988). When natural insect prey was abundant, skunks spent more time in patchy habitats, were less likely to encounter nests (which they appeared not to actively search for) and so predation rates were low anyway. Predation on upland ducks by striped skunks in areas provisioned with fish offal and sunflower seeds was over 50% lower than that on control areas (n = total 24 sites) but overall nest depredation did not change, at least in part because predation by American badgers and ground squirrels increased ('compensatory depredation', Greenwood et al. 1998), leading Greenwood et al. (1998) to suggest that supplemental food probably has limited value as a management tool because the effects can be difficult to predict (which is much like any other method of 'removing' a single predator in a system). In a review of 21 diversionary feeding studies across taxa, 13 had a direct impact on the 'problem' (i.e. a reduction in predation), but only five translated into desired management objectives (i.e. increased bird populations; Kubasiewicz et al. 2015), suggesting that suitability of the method might be species- and situation-dependent. For otters, a diversion, or 'decoy', pond (stocked with low value Crucian carp) situated between two (valuable) common carp ponds, reduced the cost of losses to the farmer because although the total loss in weight of fish was about the same as in the absence of the diversion pond, approximately a third of fish taken were of the less valuable species (Bodner 1995).

Compensation payments

In Sweden, compensation is paid to reindeer herders to cover the 'risk' of economic loss based on the number of breeding female wolverines present within a reindeer grazing district (Hobbs et al. 2012; Persson et al. 2015), whereas, in Norway, compensation is paid to herders on their ability to document losses (Tveraa et al. 2014). Both countries pay out several million dollars each year, but, in Norway, compensation is only granted for one out of four claims and there is dispute between herders and management authorities over the magnitude of losses to carnivores (Tveraa et al. 2014). In Sweden, the number of reproducing female wolverines has increased from 57 at the start of the scheme in 2002 to 125 in 2012, wolverines have started to spread into previously unoccupied areas, and herders no longer dig out dens to kill the females and her cubs (a practice that was frequent before the scheme, Persson et al. 2015). The scheme works because it protects females, the demographic segment of the population to which population growth is most sensitive, and because efficient herding (to prevent depredation) is not perversely penalized by lower compensation.

Compensation schemes for otter predation at fisheries similarly differ across Europe and there seems to be no standard approach. Schwerdtner and Gruber (2007) describe a compensation-in-advance scheme in Saxony, Germany—in which fish farmers are paid the equivalent of USD 136 annually per hectare of pond stocked with fish, for 'otter food' (see diversion ponds, above). Payment depends on otter presence (Klenke et al. 2013) and differs from other German states, where compensation (at 80% of costs) can only be claimed for damage that exceeds approximately USD 1320 in a single year and evidence (e.g. fish carcasses) must be shown to the authorities. Damage compensation is paid in the Czech Republic at an average of about USD 98 per pond, mostly (60% of claims) to hobby famers (Poledníková et al. 2013); in other countries (e.g. Portugal) no compensation is paid (Trindade 1991).

Conservation, animal welfare, ethics, and future research

The ways in which people and musteloids relate, now and in the past, are multi-faceted and cover a broad spectrum of human–wildlife interactions, modulated by the biology of the musteloids themselves, as much as by cultural differences and the changing socio-political landscapes on which they occur. Among the diverse uses and issues described in this chapter, there are two that clearly predominate: the use of musteloids for their fur, and their management for the protection of game and fisheries.

The relative smallness of musteloids means that the 'problems' encountered by people interacting with them are somewhat 'smaller' (less serious) than those encountered by people in contact with large carnivores. Musteloids are fierce predators for their size but, unlike lions (Loveridge et al. 2010), they do not kill you, your child, or (with the exception of wolverines in Scandinavia) your livestock, and, whilst musteloids may be labelled by some as a 'pest', they do not suffer the extent of cultural animosity that canids do (Sillero-Zubiri et al. 2004). Consequently, felid- and canid-related conflict more often results in retaliation by local people, demands more urgent resolution, and tends to receive more attention from the public, researchers, policy-makers, and conservationists. Likewise, whilst the exploitation of large carnivores is represented by high-profile use of a few species—wolves (for their fur), lions (for trophy hunting), tigers (for traditional medicine)—over half of all musteloid species globally are trapped as 'furbearers'. The elusivity of these small predators means that they are often little-known: few people could correctly identify, for example, a sable, and probably few that would care know that hundreds of thousands are killed each year to make coats; probably everyone can identify a lion, and the killing of just one in the summer of 2015 resulted in global mass media outcry (Macdonald et al. 2016).

There are clearly lessons to be learnt from the use and management of the better-researched (but more problematic and potentially more controversial) large carnivores, but there are also differences. Judging musteloids on their own merits, the key question is whether use, persecution, or management of musteloids (as it is currently practised) matters. Does it pose a risk to their conservation status, or affect their welfare? Is it morally acceptable?

For the most part, the fur trade and game management does not affect threatened species of terrestrial musteloids (although there are exceptions). However, there are concerns regarding sustainability, even for the non-threatened species. The impacts on musteloid populations are difficult to generalize. Large musteloids with low natural productivity tend to be vulnerable to additional sources of mortality; while small musteloids with high reproductive rates are usually resilient to losses, or they are quickly countered by immigration. But these commonalities are also influenced by the spatial scale at which populations are managed or exploited, relative to the species dispersal ability. In a review of the impacts of control on mesopredators in the US, Connor and Morris (2015) found that whilst removals of the larger species (raccoons) did not decline over time (i.e. control had no impact on population size), contrary to expectation, those of small predators (weasels and striped skunks) did. These authors caution that repeated control over large geographical areas may risk long-term population suppression for small musteloids with short dispersal distances. In fragmented landscapes, even raccoons are slow to repopulate: three years after removing raccoons from 30 forest patches (3–12 ha) in an agricultural area only 40% had returned to pre-removal densities (Beasley et al. 2013).

Otters present a more serious problem, insofar as a number of severely threatened species are both traded for their fur and persecuted (to reduce predation on fisheries), particularly (but not exclusively) in Southeast Asia. For most otter species, trade and persecution are two factors in a suite of simultaneous pressures, associated with an increasing human population that otters come into increasing contact with, in a situation that is unsustainable for both. Like lions and tigers, these depleted threatened species are unable to withstand significant additional losses and the situation urgently requires solutions.

Regardless of conservation status, the methods used to kill animals (for the fur trade or game management) are sometimes cruel. Welfare standards are inconsistent at national and international levels (Iossa et al. 2007) and there is little authoritative oversight (e.g. Born Free 2011). The Agreement on International Humane Trapping Standards (Council Decision 98/142/EC) between the European Community, Canada, and the Russian Federation (and a similar agreement with the US, Council Decision 98/487/EC) attempts to ensure adherence to acceptable methods by all fur trapping countries, for a list of species that excludes, for example, American mink and Siberian weasels. In North America, over half of fur trappers are not aware of, or do not follow, Best Management Practices (AFWA 2015) and, for some species, even these government-endorsed methods ('designed to address animal welfare') are questionable: for example, drowning semi-aquatic mink and otters in underwater foothold traps, which can be slow and inhumane (Iossa et al. 2007). Non-lethal control for game management may be preferable but, as already discussed, is not stress-free. For less common uses of musteloids—as pets, or food—there are also significant welfare issues. Wild animals destined for overseas pet shops suffer stress (and many die) during transport (Baker et al. 2013) and even the most well-meaning owners may not fully understand how to care for them or be aware of the disease risks that they bring (e.g. Harris et al. 2011). The practice, in Chinese wildmeat restaurants, of keeping red pandas (or any other mammal listed on the menu) alive in cages seems somewhat barbaric and should surely be discouraged.

Peoples' views on whether it is right to kill animals for their skins (which, these days, is for fashion rather than warmth), to protect other animals that we want to hunt, or even just for fun, depend on personal and cultural values. That nature has intrinsic value (value 'for its' own sake' or 'in its own right') is widely acknowledged, but this does not necessarily mean that killing an animal is always or necessarily wrong, only that one must have a compelling reason for doing so (Vucetich et al. 2015). The question then is what provides a 'sufficiently compelling reason'. Vucetich and Nelson (2014) conclude that wolf hunting in the US in the present day is inappropriate, and for felids, wearing spotted cat fur has become socially unacceptable (Loveridge et al. 2010). For musteloids, these types of issues are given little attention.

It is well established that (for a number of reasons already discussed) lethal control is not always effective, but, despite a move within the field of conservation biology towards a more compassionate approach (e.g. Ramp and Bekoff 2015), there is comparatively little information on the efficacy of non-lethal alternatives. Lack of monitoring, and/or data, was a general theme in our review. Scarce information on population size and status of many musteloid species makes any attempt to measure the impacts of people on musteloids challenging at best. For wolverines in Norway, for example, there is a risk that harvest quotas (to reduce predation) are based on over-estimates of population size, and the impacts of removing adult males on social structure and the risk of infanticide are unknown (Landa et al. 2000). Little quantitative data are available on the extent, frequency, cost, or consequences of conflict. Like the canids that conflict with humans, management of non-threatened musteloids is required, but, unlike the management of canids (for which there is a considerable body of research, reviewed in Sillero-Zubiri et al. 2004), experimental tests of the effectiveness of non-lethal solutions for musteloids are rare.

With increasing urbanization of natural areas, urban musteloids (usually common species that are able to tolerate human disturbance and adapt to utilize human resources and subsidies) are an increasing problem. Proposed solutions, in this case, need to take account of the well-being of the people that live there, but it is important to recognize that urban wildlife also has benefits (for example, skunks prey on damaging insects, Dragoo 2009b) and brings pleasure to many other people (Gehrt 2004; Rohde and Kendle 1994). Many of the problems associated with urban species can be reduced by modifying human behaviour (preventing access to buildings, and reducing the availability of human subsidies—securing rubbish, removing fallen fruit and pet food—that allow some urban species to attain artificially high densities), which may in any case be a more effective strategy than managing the problem post hoc (Gehrt 2004) and falls into the kind of holistic ecological solution advocated by Ramp and Bekoff (2015). Currently these urban populations of musteloids are probably not important for the global conservation status of the species but increasing urbanization of the planet means that they may be in the future (e.g. Cypher et al. 2010).

In conclusion, the trade of colossal numbers of furs dominated musteloid–people relationships in the past, with consequential welfare and ethical issues that are still relevant today. The management of musteloids considered 'pests' (by some) presents challenges that are biological (do current control methods actually work?) as much as ethical. Emerging issues such as sharing urban spaces and pet traffic pose questions not formerly considered. The general lack of systematic quantitative information means that we struggle at the current time to properly evaluate the conservation impacts of the use or management of musteloids, or the efficacy of alternative (non-lethal) methods. Considerable effort is expended in protecting game birds, considerably less seems to be expended in assessing the effects of our efforts on the musteloids themselves. Although few are considered to be 'threatened', many are declining, and the devastation of the past suggests that there is little room for complacency.

Acknowledgements

Thanks to Pat Morris, Sam Zeveloff, and Andrew (Harry) Harrington for supplying photos, and to David Macdonald and Chris Newman for comments on an earlier version.