

PART 3

Major issues in Canid Conservation

Canid Sociology – a Brief Overview

D.W. Macdonald

10.1 Introduction

Canids demonstrate enormous diversity in appearance and lifestyle. This variation, and particularly the salient themes in canid behavioural ecology, is the topic of six substantial reviews and 14 detailed case studies in *The Biology and Conservation of Wild Canids* (Macdonald and Sillero-Zubiri 2004a). In particular, Macdonald *et al.* (2004) summarise the nature of canid societies, and discuss the selective pressures that have fashioned them. Here, we provide a brief synopsis of this aspect of canid biology, with the purpose of ensuring that those planning their conservation are alert to the relevant features of their societies.

10.2 Canid diversity

Canid species vary in size, distribution, diet, habitat and home range, all of which are reflections of diverse adaptations. The impressiveness of their interspecific diversity in behaviour (previous reviews include Macdonald and Moehlman 1982; Creel and Macdonald 1995) is almost matched by the extent of intraspecific variation in their biology (e.g., Englund 1970; Macdonald 1981). However, for all the diversity that exists amongst canids, differences are merely variations upon the strikingly consistent themes of canid biology, notably their opportunism and versatility, their territoriality, and their societies built from a foundation of behavioural monogamy with its attendant intrasexual dominance hierarchies, social suppression of reproduction and helpers (Macdonald and Sillero-Zubiri 2004b).

Formerly, wild canids were thought of as approximating one of two models. Either they were small, ate small prey and lived essentially solitary lives, as was supposed to be the model of the red fox (*Vulpes vulpes*). Alternatively, they were large, ate large prey and used the collective power of a pack to hunt, catch and defend it. The idea that canid society was a reflection of their food (and other resources) was well founded, but the simplicity of this early distinction was not. This is clearly illustrated by the mixture of attributes of the Ethiopian wolf (*Canis simensis*). As summarised by Sillero-Zubiri *et al.* (2004a), this canid is relatively large, eats almost exclusively small prey, lives in packs but mostly hunts alone. Furthermore, it became clear that while red foxes did live in territorial pairs in some circumstances,

elsewhere they formed groups (Macdonald 1987) and, conversely, although wolves do often live in packs and kill large prey, some individuals and some populations do not, at least in some seasons (e.g., Mech and Boitani 2003). In short, the adaptive significance of canid society required more explanation than initially supposed.

Canids are highly communicative. Studies of their postures (e.g., Golani and Keller 1975), vocalisations (e.g., Tembrock 1962; Harrington 1987; Frommolt *et al.* 2003) and scent marking (e.g., Peters and Mech 1975; Bekoff 1978; Macdonald 1980c; Sillero-Zubiri and Macdonald 1998), all reveal subtleties of communication that make it clear that their societies must be complex. An overview allows their societies to be classified, very broadly, into three categories, which differ in terms of body weight, group size, group composition and dispersal behaviour. Amongst mammals in general, dispersal distances tend to be greater in males than females and, where there is a skew, males tend to be the dispersing sex (Greenwood and Harvey 1982). Macdonald and Moehlman (1982) and Moehlman (1986, 1989) noticed that this generalisation did not apply uniformly throughout the canids. In brief, small canids (<6kg) tend to live in pairs or, when they form groups these tend to female-biased sex ratios, young males tend to emigrate, and females stay in their natal range as helpers (most species of small vulpine canids are examples). Medium-sized canids (6–13kg) have an equal adult sex ratio and emigration rate, and both sexes may be helpers – examples include three species of jackal and coyotes (*Canis latrans*), and some of the South American canids such as the crab-eating fox (*Cerdocyon thous*) (Macdonald and Courtenay 1996). Large-sized canids (>13kg) exhibit an adult sex ratio skewed towards males, female emigration and male helpers. Notwithstanding variation between populations (see McNutt 1996a), examples would be African wild dogs (*Lycaon pictus*), dholes (*Cuon alpinus*) and Ethiopian wolves. However, this simple classification is no more than a guide – for example, the bush dog (*Speothos venaticus*) is rather small, but fits the third behavioural model (Macdonald 1996), whereas the maned wolf (*Chrysocyon brachyurus*) is large and does not (Dietz 1984).

10.3 Selection for sociality

With this variation in canid societies, what might determine

the way a particular species, or population of a species, behaves? Creel and Macdonald (1995) summarised five general families of selective pressure, or conditions, which may select for sociality in carnivores generally. Two of these selective pressures reduce the costs of tolerating conspecifics:

1. *Resource dispersion*: most obviously, abundant prey, rich or variable prey patches, or rapid prey renewal, may all lead to low costs of tolerating conspecifics;
2. *Dispersal costs*: constraints on dispersal opportunities such as lack of suitable habitat, low mate availability, or intraspecific competition may favour the retention of young within their natal group past the age of maturity. In short, the balance of these two essentially ecological factors may facilitate group formation, respectively by making the costs of doing so minimal and the costs of not doing so high. In contrast, the balance of three essentially behavioural selective pressures may increase the benefits of tolerating conspecifics (up to some optimum group size);
3. *Resource acquisition*: groups may use strength of numbers in the acquisition and retention of resources, e.g., hunting in groups may increase foraging success, where prey are large or difficult to kill, groups may also fare better in territorial defence and intra- and inter-specific competition for food, especially at large kills;
4. *Defence against predation*: groups may be less vulnerable to predation or attack; and
5. *Reproductive advantages*: group membership offers the opportunity for shared feeding and protection of young.

It is clear that various different types of society would result depending on the balance of these five categories of selective pressure (as explored by Macdonald and Carr 1989). For example, all else being equal, group size is likely to be less where dispersal opportunities are greater, whereas cub survival to weaning might be expected to increase in the presence of allo-parents.

These outcomes are relevant to conservation planners insofar as the prey base, opportunities for dispersal or recruitment, and patterns of grouping will affect the viability of a population and the options for its conservation.

What evidence exists that these five categories of selective pressure do affect canid societies? Consider first the three behavioural selective pressures for group living.

10.3.1 Behavioural selective pressures for sociality

Strength in numbers

The notion that canids hunt together in order to more effectively overwhelm prey too challenging to be hunted alone is intuitively plausible. However, it has proved

extremely difficult to support empirically. Early data on black-backed jackals (*Canis mesomelas*) (e.g., Wyman 1967; Lamprecht 1978) proved inconclusive. Even with the most conspicuous pack-hunters, African wild dogs, evidence that individual pack members did better when hunting in larger groups proved equivocal when the measure of success was the quantity of prey eaten (Fanshawe and Fitzgibbon 1993). Indeed, a review of wolf hunting success by Schmidt and Mech (1997) revealed a general decline in food intake per wolf per day with larger pack sizes, and Vucetich *et al.* (2004) illustrate that in the absence of scavengers, wolves would do better hunting in pairs not packs. However, when Creel and Creel (1995) shifted the focus from the bulk of prey eaten to the profit and loss account of catching them, it became more convincing that wild dogs hunting in large groups may provide a net benefit to participating individuals (Creel 1997; Creel and Creel 2002).

Intraspecific interference competition at kills can be considerable and the outcome of such competition generally appears to be affected by group size. Irrespective of the contribution of collaboration to making a kill, larger groups of coyotes emerged as more successful at defending kills (Bekoff and Wells 1982). Interspecific interference competition is similarly a major component of canid ecology, with a sequence of species competing to retain kills (Mills 1989a, 1989b). It is not clear what function of aggregate body weight and numbers of sets of teeth determines the algebra of victory when a larger number of smaller carnivores battles with a smaller number of larger ones, but it is clear that strength of numbers is important when, for example, wild dogs strive to repel hyaenas from a kill (McNutt 1996a), or grey wolves (*Canis lupus*) seek to keep scavenging corvids at bay (Vucetich *et al.* 2004). Strength of numbers may also be important in territorial defence, and evidence accumulates that larger groups are more successful at defending territories (e.g., Bekoff and Wells 1982; Creel and Creel 1998). A large pack of golden jackals (*Canis aureus*) habitually stole food from a smaller pack (Macdonald 1979a), and amongst Ethiopian wolves, larger packs invariably prevail in territorial clashes (Sillero-Zubiri and Macdonald 1998). Territorial clashes may account for a substantial proportion of adult mortality (e.g., Mech 1977; Creel and Creel 1998).

Predation

There are few data from canids to support the intuition that larger groups are more vigilant, having more pairs of eyes, ears and nostrils with which to detect predatory danger more rapidly. Wild dogs will aggressively mob potential predators if pups are threatened (e.g., Kühme 1965; Estes and Goddard 1967) and this becomes more effective for larger packs. Larger groups of bat-eared foxes (*Otocyon megalotis*) appear better able to repel

predators approaching their dens (Maas and Macdonald 2004). The magnitude of predation as a force in canid lives is illustrated by the fact that lions are responsible for 43% of the natural mortality of wild dogs in Kruger National Park (Mills and Gorman 1997; see also Carbone *et al.* 1997).

Reproductive advantages

Canid pups have a prolonged period of dependency on adults and are commonly tended by both parents (Kleiman and Eisenberg 1973). Lupine canids are the only Carnivore group to regurgitate, and males also provision pregnant and lactating mates allowing them to better direct energy into gestation and lactation, and also to guard young uninterruptedly (Ofstedal and Gittleman 1989). The original list of species for which non-breeding ‘helpers’ fed and tended the young (e.g., red foxes, black-backed jackals, grey wolves – Macdonald 1979b; Moehlman 1979; Fentress and Ryan 1982) has expanded with the number of species studied, revealing allo-parental care as a widespread trait of the canids (e.g., Bekoff and Wells 1982). While allo-parental care appears self-evidently helpful (Moehlman 1979), demonstration that helping translates into improved reproductive success has also proven difficult. Perhaps the most extreme form of allo-parental behaviour is allo-suckling, observed in, for example, red and Arctic foxes (*Alopex lagopus*), grey and Ethiopian wolves, coyotes and bat-eared foxes. Various mechanisms may lead to allo-suckling: a subordinate female may lose her pups through the dominant’s infanticide (van Lawick 1974; Malcolm and Marten 1982; Corbett 1988b), reproductive suppression may fail leading to mixed litters, and some

females appear to lactate spontaneously (Sillero-Zubiri *et al.* 2004a). Evidence that allo-suckling benefits the pups has proved elusive (Malcolm and Marten 1982; Zabel and Taggart 1989; Roulin and Heeb 1999), perhaps because they materialise only in the long term (Emlen 1991) making pup survival to weaning or dispersal an inappropriate measure. Similarly, when it comes to measuring the consequences of any form of allo-parental behaviour, a plausible measure might be the life-time reproductive success of the mother rather than the survival of a particular litter of offspring – the underlying mechanism being that the helpers’ contribution reduced the energetic burden on the mother, thereby extending her subsequent performance. However, long-term life-history studies with the capacity to test for such effects are few.

Canid society is typified by reproductive suppression exerted by dominant females on their subordinates (see Moehlman 1986, 1989) and has been recorded in at least 44% of 25 species for which there is information (Moehlman and Hofer 1997). The degree of suppression, however, varies both intra- and interspecifically (Creel and Waser 1991, 1994) and subordinates do reproduce, albeit at lower rates than dominants (Packard *et al.* 1983; Macdonald 1987; Fuller and Kat 1990). Although behavioural suppression appears to be common in males (e.g., Packard *et al.* 1983), multiple paternity has been reported (Gottelli *et al.* 1994), and the likelihood is that a combination of behavioural and genetic techniques will increasingly reveal that complications in mating systems are widespread. Various mechanisms may be involved in social suppression of reproduction (Packard *et al.* 1985), but whatever the mechanism, the outcome is that amongst



Páll Hersteinsson weighs an Arctic fox cub in Hlõðuvík Bay after luring it out of the den by imitating the parents’ call and ear-tagging it. Hornstrandir Nature Reserve, Iceland.

Hólmfríður Sigþórsdóttir

grey wolves, subordinates rarely have offspring and generally lose those they do have (Peterson *et al.* 1984). Nonetheless, in Kruger National Park, 40% of 25 dens contained pups of more than one female wild dog (although only 9% of pups were the offspring of subordinates) (Girman *et al.* 1997).

The benefits provided by helpers may vary according to the sex of the helper; clearly, only females have the potential to allo-suckle, but less obviously, data on grey wolves and wild dogs indicate that males provide more solid food to pups per capita than do females (Fentress and Ryon 1982; Malcolm and Marten 1982).

In short, membership of larger groups may bring canids advantages due to combinations of improved foraging efficiency, breeding success, and survivorship. In some cases, and it has been argued for wild dogs, the advantages of sociality may be reflected as an Allee effect: a positive feedback loop of poor reproduction and low survival culminating in failure of the whole pack (Courchamp and Macdonald 2001; Courchamp *et al.* 2001). In so far as pack sizes reflect some optimum of different selective pressures, one would expect a dome-shaped distribution of pack sizes, with populations subject to inverse density dependence at low density and direct density dependence at high density, exactly as observed by Creel (1997) for wild dogs.

10.3.2 Ecological selective pressures for sociality

While behavioural benefits may be amongst the selective pressures favouring sociality in wild canids, ecological factors create the framework within which these pressures operate, and dictate the balance of costs and benefits between group membership and dispersal. Alexander (1974) was the first to suggest that group formation (and cooperative behaviour between their members) is a secondary consequence of group living that is initially favoured by some other ecological reason. One crucial ecological factor is resource dispersion.

Resource dispersion

Irrespective of the advantages of group living, resource dispersion may significantly affect the costs of grouping. The idea that certain patterns of resource availability might facilitate group formation by making coexistence feasible, grew especially out of observations on badgers (*Meles meles*) (Kruuk 1978), and was formalised as the Resource Dispersion Hypothesis (RDH) by Macdonald (1983) and Carr and Macdonald (1986). The hypothesis is that groups may develop where resources are dispersed such that the smallest economically defensible territory for a pair can also sustain additional animals. The RDH offers an explanation of group size variance, regardless of whether individuals gain from each other's presence or

not. Not only may it apply to current societies, but it may describe the conditions that favoured the evolution of sociality. While RDH can only be tested by manipulative experiments (Johnson *et al.* 2002), several field studies of canids are broadly in line with its predictions (Hersteinsson and Macdonald 1982; Macdonald 1983; Geffen *et al.* 1992c; but see Baker and Harris 2004).

Dispersal

Dispersal is crucial both to understanding life-history processes (Waser 1996) and to conservation (Macdonald and Johnson 2000). However, it is poorly understood, and philopatry brings with it the risk of inbreeding. Examples of neighbourhood relatedness and return from dispersal are mounting amongst canids and may be widespread (Lehman *et al.* 1992; Macdonald and Courtenay 1996). Inbreeding may also occur locally between related pack founders derived from neighbouring packs (Mech 1987). Messier and Barrette (1982) conclude that the most influential factor leading to group formation in coyotes is the absence of dispersal opportunities. The ethology of dispersal is correspondingly poorly understood and is often reported as a summary bee-line distance. However, such distances often disguise a more complicated reality (Macdonald 1980d). The time between emigrating and settling can be highly variable (Gese and Mech 1991). Some animals mate while on forays but never actually settle in that group, and coalitions of females may also leave their natal range as a group to settle in previously unoccupied territory, where they are joined by an emigrating male (Gese and Mech 1991). Although Mech's (1987) study of wolves illustrates almost every variant imaginable, some generalisations may emerge – for example, Macdonald and Bacon (1982) noted that the mean bee-line distance of red fox dispersal correlated with territory sizes, with the result that foxes travelling very different distances were crossing rather similar numbers of territories.

10.4 Trends in canid sociality

Extending the broad association between body size and social system, Moehlman (1986, 1989) reported that female body mass was positively correlated with gestation length, neonate mass, litter size and litter mass, and that from these corollaries of size she deduced generalisations about interspecific differences in adult sex ratio, dispersal, mating, and neonate rearing systems. Female body weight was argued to be the ultimate driver of a cascade effect such that relative to smaller species, larger canids invest more in prepartum reproduction (having larger neonates and a large number of offspring), live, and hunt, in larger groups, generally prey on larger, vertebrate prey, and have a greater incidence of reproductive suppression and allo-

parental behaviour (Moehlman and Hofer 1997). Diet breadth, maximum prey size and incidence of cooperative hunting all scale allometrically with body size in canids. Amongst carnivores as a whole, litter mass, litter growth rate and total investment are higher in communally breeding species with reproductive suppression (Gittleman 1986; Creel and Creel 1991).

In a similar comparative analysis, Geffen *et al.* (1996) found a high correlation and isometric relationship between neonatal weight and female body weight and interpreted this as evidence that the size of newborn canids is constrained either by female body size directly or by some allometric correlate of female body size e.g., pelvic width (Leutenegger and Chaverud 1982). Their analysis, unlike Moehlman's (1986), took account of phylogeny, and found litter size to be only weakly and non-isometrically correlated with female body weight, suggesting that litter size may be adjusted in response to the availability of resources. They suggested also that female pre-birth investment can only be adjusted by varying litter size – red and Arctic foxes, and wolves are amongst canid species exhibiting decreases in litter size with decreases in prey abundance (Macpherson 1969; Harrington *et al.* 1983; Lindström 1989; Angerbjörn *et al.* 1991; Hersteinsson and Macdonald 1992). On this view, changes in body size, litter size and social organisation within the Canidae may be attributed primarily to differences in food availability. Thus, small canids (e.g., fennec fox *Vulpes zerda*) are usually associated with arid and poor habitats in which only a small body mass can be supported year round, whereas large canids are often associated with habitats in which prey are at least very abundant (e.g., Ethiopian wolves) and more generally, abundant and large (e.g., African wild dogs, grey wolves).

The relationship between group size and territory size is intriguing. As Kruuk and Macdonald (1985) noted, starting from a minimum defensible territory there are two possibilities for group formation. First, contractionism or building up a group by bringing extra members into the minimum territory that will support a pair. Second, expansionism is the alternative, where the benefits of sociality may be so great, that it pays the group to expand to a bigger territory. Insofar as group size and territory size may be uncorrelated in red foxes, but strongly correlated in Ethiopian wolves, there is evidence for both scenarios. Furthermore, an analysis of group metabolic requirements plotted against home range sizes reveals that some group-living canids have smaller home ranges than predicted by their collective body weight, whereas for others the group home ranges are larger than predicted (Johnson *et al.* in press). Macdonald *et al.* (2004) argue that RDH may explain both negative and positive deviations by social canids from the home range size predicted by their metabolic needs. In both cases ecological circumstances create conditions that diminish or obliterate the costs of group formation: a patch of invertebrates may allow several canids to feed together, as may the body of a single large ungulate; groups of canids feeding on the former tend to be associated with smaller than expected territories, groups of canids feeding on the latter tend to be associated with larger ranges than expected. In both cases, it will be a matter of local, autecological circumstances whether a particular species or population opts to form groups up to the size potentially accommodated in such sharable enclaves, or even to form larger groups requiring expansionist territories, or not to share at all.

Conservation Genetics of Canids¹

R.K. Wayne, E. Geffen and C. Vilà

11.1 Introduction

In this chapter, population genetic data for canid species that have recently been studied with molecular genetic techniques, and that warrant conservation concern, will be reviewed. A brief summary of the pertinent findings for each studied species is followed by specific conservation implications. For a full discussion of findings on these canid species and others, please see Wayne *et al.* (2004) and Wayne (1996).

In general, because canids are highly mobile carnivores, levels of genetic differentiation are low; however, habitat fragmentation and loss has caused some populations to become isolated and genetic drift in these small populations has accelerated differentiation and loss of genetic variation. Moreover, genetic loss has occurred in some endangered species because of persistent small population sizes. Also, in many species, a history of dynamic changes in abundance and distribution are superimposed on current demographic conditions. For example, genetic analysis of grey wolves (*Canis lupus*) and coyotes (*C. latrans*) has suggested range contractions during glacial maxima followed by reinvasion across several continents (Fрати *et al.* 1998; Vilà *et al.* 1999).

High mobility also influences the degree to which interspecific hybridisation affects the genetic composition of hybridising species (Lehman *et al.* 1991; Jenks and Wayne 1992; Wayne 1992; Mercure *et al.* 1993; Wilson *et al.* 2000; Wayne and Brown 2001). The width of a hybrid zone is a function of the distance travelled from birth to the place of first reproduction and the degree of natural selection against hybrids (Barton and Hewitt 1989). If selection is weak, hybrid zones may span a considerable distance in highly mobile species and interspecific gene flow may strongly affect the genetic heritage of hybridising forms (Jenks and Wayne 1992; Wilson *et al.* 2000; Wayne and Brown 2001). Interspecific hybridisation contributed to the genetic extinction of red wolves (*C. rufus*) in the wild, has greatly compromised the genetic composition of the Great Lakes wolf (*C. lupus lycaon*) and has flooded New England with wolf-coyote hybrids (Nowak 1979; Lehman *et al.* 1991; Jenks and Wayne 1992; Roy *et al.* 1994b, 1996; Wilson *et al.* 2000; P. J. Wilson *et al.* unpubl.). Further, hybridisation between domestic dogs and

Ethiopian wolves (*C. simensis*) (Gottelli *et al.* 1994) and perhaps between grey wolves and domestic dogs in certain areas may be consequential (Vila and Wayne 1999; Anderson *et al.* 2002; Randi and Lucchini 2002; Vilà *et al.* 2003a).

11.2 Review of studies on rare and endangered canids

11.2.1 Ethiopian wolf

The Ethiopian wolf is one of the most endangered living canids (Gottelli and Sillero-Zubiri 1992; Sillero-Zubiri and Macdonald 1997; Marino 2003). The total population in 2000 was less than 500 individuals. The species is dispersed across the Ethiopian highlands above 3,000m a.s.l. in small, highly isolated populations (Gottelli and Sillero-Zubiri 1992; Marino 2003). Phylogenetic analysis of mtDNA sequences showed that the closest living relatives of Ethiopian wolves are probably grey wolves and coyotes (Gottelli *et al.* 1994, 2004; Vilà *et al.* 1999). An evolutionary hypothesis consistent with these results is that Ethiopian wolves are a relict form remaining from a Pleistocene invasion of a wolf-like progenitor into East Africa. The current extent of Ethiopian high altitude moorland habitats is only 5% of the area existing after the last Ice Age (Yalden 1983; Gottelli *et al.* 2004). Consequently, the geographic range and numerical abundance of Ethiopian wolves has likely decreased during the Holocene. More recently, habitat loss and fragmentation due to human population growth and agriculture have accelerated the decline of Ethiopian wolves.

Genetic variation, differentiation and hybridisation

Mitochondrial DNA (mtDNA) sequence analyses and microsatellite typing showed that the two populations in the Bale Mountains had very low variability. All wolves had the same mitochondrial DNA haplotype, and microsatellite loci was only 46% and mean allelic diversity 38% of that commonly found in other wolf-like canids. Such low levels of heterozygosity are consistent with an equilibrium effective population size of only a few hundred

¹ This paper is condensed from Wayne, R.K., Geffen, E. and Vilà, C. 2004. Populations and conservation genetic of canids. Pp. 55–84 in D.W. Macdonald and C. Sillero-Zubiri, (eds). *Biology and conservation of wild canids*, Oxford University Press, Oxford, UK.

individuals (Gottelli *et al.* 1994). A recent analysis of mtDNA from six other populations identified genetic differences but found low number of haplotypes and a low sequence divergence among them, confirming the wolves' recent evolution (Gottelli *et al.* 2004). The genetic structure observed was congruent with random fixation of alleles within isolated populations as habitat contracted since late Pleistocene.

Although loss of variation and inbreeding in isolated populations are concerns for endangered species, an additional problem for Ethiopian wolves is hybridisation with domestic dogs. Genetic analysis showed that suspected hybrid individuals in a population in the Sanetti Plateau of the Bale Mountains National Park had microsatellite alleles not otherwise found in Ethiopian wolves, but were present in domestic dogs. In contrast, these individuals had mtDNA haplotypes identical to those in 'pure' Ethiopian wolves (Gottelli *et al.* 1994), a result consistent with field reports that interspecific matings only involved male domestic dogs and female Ethiopian wolves (Sillero-Zubiri 1994). Additionally, parentage analysis found that a single litter had both a wolf and dog as fathers, showing that multiple paternity occurs in wolves and can involve both species. Dogs not only hybridise with Ethiopian wolves and compete with them for food, but also are reservoirs of canine diseases (Sillero-Zubiri *et al.* 1996b).

Conservation implications

The sharply lower levels of variation in the Ethiopian wolf reflect a long history of population declines compounded by recent habitat fragmentation (Gottelli *et al.* 2004). However, perhaps a greater concern than the reduced levels of genetic variation is the vulnerability of the few remaining populations to diseases such as rabies, which is already thought to have eliminated about one-half of the Bale Mountains population (Sillero-Zubiri *et al.* 1996b; Sillero-Zubiri and Macdonald 1997) and to other stochastic demographic effects. Inbreeding depression may occur in canids (Laikre and Ryman 1991; Laikre *et al.* 1993; Fredrickson and Hedrick 2002; but see Kalinowski *et al.* 1999) and may conceivably influence the persistence of the population (e.g., Seal and Lacy 1998). Loss of genetic variation in small populations may also influence the ability of the population to adapt to changing conditions (Frankham *et al.* 2002). However, Ethiopian wolves actively avoid inbreeding (Sillero-Zubiri *et al.* 1996a) thus decreasing the rate at which genetic variation is lost and mitigating the effect of inbreeding. In one population, the loss of unique Ethiopian wolves' characteristics may result from interbreeding with dogs. However, this threat may be restricted to that locality (Wayne and Gottelli 1997). Ethiopian wolves are not being bred in captivity (Sillero-Zubiri and Macdonald 1997) and the genetic results suggest that a reservoir of pure wolves must be protected and bred in a captive setting as a source for reintroduction, should

efforts to sustain the wild population fail. Finally, genetic surveys of other populations of Ethiopian wolves should continue so that a balanced programme of captive and *in situ* management can be constructed that maintains historic levels of variation within, and gene flow between, populations (Wayne and Gottelli 1997; Crandall *et al.* 2000; Gottelli *et al.* 2004).

11.2.2 African wild dog

The African wild dog (*Lycaon pictus*) once ranged over most of sub-Saharan Africa, inhabiting areas of dry woodland and savannah (Woodroffe *et al.* 1997). However, due to habitat loss, hunting and disease, many populations have vanished or are severely reduced in number. The extant populations are highly fragmented and total no more than several thousand individuals (Fanshawe *et al.* 1997; Ginsberg and Woodroffe 1997a). Importantly, the western African and Kenyan populations are nearing extinction, yet these populations are not represented in zoos as only South African wild dogs are kept in captivity. Populations in South Africa currently are stabilised in protected areas (Fanshawe *et al.* 1997).

Genetic variation and population differentiation

Analysis of mtDNA nucleotide diversity suggested that wild dog populations have historically been small relative to other large carnivores (Girman *et al.* 2001). However, recent population declines due to human induced habitat loss have not caused a dramatic reduction in genetic diversity. Levels of diversity in microsatellite loci do not show strong evidence of recent or historic population decline relative to other carnivores. Further, the levels of genetic polymorphism estimated from the microsatellite data were relatively similar in all seven populations. Although the average sample size for each population varied greatly (5.8 in the North-west Namibia population to 93.8 in the Kruger population, South Africa), the mean number of alleles per locus ranged only between 3.4 and 4.4. Heterozygosity values were also similar, ranging from 0.56 for the Kruger population to 0.67 for the Selous population. The heterozygosity of a captive South African population was lower (0.50) and the mean number of alleles per locus was only 3.3.

Mitochondrial and microsatellite loci showed significant differentiation between populations. Eastern and southern populations may have been historically isolated. One historic and eight recent mtDNA haplotypes were found that defined two highly divergent clades. In contrast to a previous more limited mtDNA analysis (Girman *et al.* 1993), sequences from these clades were not geographically restricted to eastern or southern African populations. Rather, a large admixture zone was found spanning populations from Botswana, Zimbabwe and south-eastern Tanzania. Genetic differentiation between populations

was significant for both microsatellite and mitochondrial markers and unique mtDNA haplotypes and alleles characterised the populations. However, gene flow estimates (Nm) based on microsatellite data were moderate to high in the range of 1.53 to 5.88 migrants per generation. In contrast, gene flow estimates based on the mtDNA control region were lower than expected in the range of 0.04 to 2.67 migrants per generation. Given the differences in the mode of inheritance of mitochondrial and nuclear markers, the results suggest a male bias in long distance dispersal. However, dispersal distance has been found to be similar for males and females in a Botswana population (McNutt 1996a), so the genetic results could indicate a higher frequency of male dispersal. Past and present distribution of the *miombo* (*Brachystegia-Julbernardia*) woodland and grassland, as well as the barrier imposed by the rift valley are biogeographic factors that may explain the current distribution of genetic variability (Girman *et al.* 2001). However, West African populations, represented by a single sample from a museum specimen, define a distinct branch suggesting a history of genetic isolation.

Conservation implications

Two haplotype clades co-occur over much of the current geographic range of the wild dog, which likely reflects natural mixing of previously isolated populations (Girman and Wayne 1997; Girman *et al.* 2001). Consequently, genetic management should aim at mimicking observed levels of gene flow between contiguous populations within this admixture zone (Crandall *et al.* 2000; Wayne and Brown 2001). Individual-based models of wild dog population dynamics suggest that even low rates of migration between populations can demographically stabilise populations otherwise at risk of extinction (Vucetich and Creel 1999). However, in wild dogs, genetic differentiation of microsatellite loci increases with distance and eastern and southern African populations may be morphologically distinct. Consequently, translocations between geographically distant southern and eastern populations are not advised because adaptive differences may exist (Crandall *et al.* 2000). For the Masai Mara and Serengeti, where wild dogs are threatened or even locally extinct (Fanshawe *et al.* 1997), the Selous region would be an appropriate source of individuals for reintroduction at the level of a few migrants per generation (see above). Additionally, because the genetic results suggest more frequent dispersal and/or longer dispersal distances in males than in females, the population management strategy should focus on the more frequent translocation of males to replicate natural processes. West African populations should be a high priority for research and conservation, especially given evidence of genetic distinction and their perilous population status.

Finally, to ameliorate genetic decline, population sizes should be kept as large as possible given the remaining

habitat area. Additionally, gene flow should be facilitated by maintaining corridors that link populations, and, when this is not possible, through translocation at historic levels as indicated by genetic data. The maintenance of genetic variation, especially the component that influences fitness, is critical to population persistence and the future evolutionary response of wild dogs to changing environmental conditions (Crandall *et al.* 2000).

11.2.3 Grey wolf

The grey wolf has the largest historical geographic range of any canid, and exists in a wide range of habitats from cold tundra to the warm deserts of the Old and New World. Because grey wolves are the most mobile canid species, genetic differentiation between populations connected by appropriate habitat is expected to be low. However, wolves vary geographically in body size and pelage suggesting selection causes differentiation despite high levels of gene flow. For example, selection for differences according to habitat type (e.g. tundra *vs.* boreal forest) or prey (migratory *vs.* resident, large *vs.* small; Kolenosky and Standfield 1975; Peterson *et al.* 1998; Carmichael *et al.* 2001) could presumably cause differentiation despite gene flow. Wolf populations need to be connected by viable corridors. Despite the high potential mobility of wolves, habitat fragmentation and habitat loss can dramatically affect the demography and genetic variability of wolf populations. For example, western European populations have reduced mitochondrial DNA variation within populations but often have unique mitochondrial DNA haplotypes (Wayne *et al.* 1992; Randi 1993; Randi *et al.* 2000; see below). Similarly, by reducing the effective population size of isolated populations, predator control programmes may cause declines in genetic variation, an increase in levels of inbreeding and a disruption of social hierarchies (Ellegren *et al.* 1996; Ellegren 1999; Vilà *et al.* 2003b). Alternatively, populations that are controlled may also become population sinks if immigration is common which may enhance genetic variation (Fрати *et al.* 1998; Wang and Ryman 2001).

Genetic variation

Genetic variability within large interconnected wolf populations is generally high. With the exception of the Italian wolf population, large populations in the Old and New World have several mtDNA control region or mtDNA RFLP haplotypes (Wayne *et al.* 1992; Vilà *et al.* 1999; Randi *et al.* 2000) and have high values of nucleotide diversity. In North America, only the Mexican wolf (*Canis lupus baileyi*) has low levels of variation (Roy *et al.* 1994b). Genealogical measures of nucleotide diversity suggest that grey wolves were more abundant than coyotes in the past and that both species declined throughout the Late Pleistocene, although wolves declined more rapidly. In

general, nucleotide diversity data imply a decline in grey wolves from over five million breeding females (about 33 million wolves) worldwide in the late Pleistocene to about 173,000 breeding females (1.2 million wolves) in the recent past. Today, less than 300,000 exist worldwide (Boitani 2003).

Dramatic demographic declines or historical population bottlenecks have been documented for some wolf populations and genetic studies have found them to contain less genetic variation. For example, the Italian wolf population declined dramatically in the 18th and 19th century due to habitat loss and predator-control programmes (Randi 1993; Randi *et al.* 1995, 2000; Scandura *et al.* 2001) and extensive mtDNA studies showed these wolves to have a single mitochondrial haplotype, which represents lower diversity than that in other Old World populations (Wayne *et al.* 1992; Randi *et al.* 1995, 2000; Vilà *et al.* 1999; Scandura *et al.* 2001). The Italian wolf haplotype is unique, and is otherwise found only in French wolves, a population recently founded by wolves from Italy (Taberlet *et al.* 1996; Lucchini *et al.* 2002; Valière *et al.* 2003). However, levels of microsatellite variation approach that in large wolf populations (Randi *et al.* 2000; Scandura *et al.* 2001).

Scandinavian wolves have declined over the past few hundred years to the point of near extinction in the 1970s. However, a new group of wolves was discovered in southern Sweden in the early 1980s which was thought to be the founding stock of the current Scandinavian population, estimated to be about 100 individuals in 2000 (Vilà *et al.* 2003b). Genetic studies suggested that the current population is reduced in genetic variation and that variability was being lost over time (Ellegren *et al.* 1996; Vilà *et al.* 2003b). The Scandinavian population has 71% of the variation in the large neighbouring population of Finland and Russia and is fixed for a single mtDNA haplotype. The level of inbreeding observed in the Scandinavian wolves is similar to that of the Swedish captive population (Ellegren 1999) in which inbreeding depression was detected (Laikre and Ryman 1991; Laikre *et al.* 1993). The southern Scandinavian population has a single control region haplotype, unique microsatellite alleles and Y-chromosome haplotypes, thereby excluding the possibility that it had been founded by individuals released from captivity (Sundqvist *et al.* 2001). Genetic data suggest that the Scandinavian population was founded by two individuals that successfully migrated from the Finnish-Russian population and established a breeding pack in 1983 (Vilà *et al.* 2003b). The arrival of a new male migrant, reproducing for the first time in 1991, allowed the temporary population recovery of the population and avoided extreme inbreeding (Vilà *et al.* 2003b).

The Mexican wolf has declined to extinction in the wild due to habitat loss and an extensive extermination programme in the first half of the 20th century. Two of the

three captive Mexican wolf populations had fewer microsatellite alleles and reduced heterozygosity (García-Moreno *et al.* 1996; Hedrick *et al.* 1997). Moreover, only two mtDNA haplotypes were found in the three captive populations (Hedrick *et al.* 1997). The total founding population numbered about seven. In the past, only the certified lineage, founded from three individuals of known Mexican wolf ancestry, was used in the captive breeding programme. However, recent genetic analysis established a close relationship among the three captive populations and found no evidence of dog, coyote or Northern grey wolf ancestry (García-Moreno *et al.* 1996; Hedrick *et al.* 1997). Consequently, to preserve the maximum genetic diversity of the Mexican wolf, plans to interbreed the three populations were developed. Like captive Swedish wolves, Mexican wolves showed signs of inbreeding depression (Fredrickson and Hedrick 2002; see also Kalinowski *et al.* 1999).

Genetic differentiation

Grey wolves show evidence of genetic differentiation on regional and continental scales. Wolves in the Old and New World do not commonly share mtDNA haplotypes (Wayne *et al.* 1992; Vilà *et al.* 1999). The degree of genetic subdivision among populations differs in wolves of the Old and New World (Wayne *et al.* 1992; Roy *et al.* 1994b; Ellegren *et al.* 1996; Forbes and Boyd 1996, 1997; Ellegren 1999; Randi 1993; Randi *et al.*, 1995, 2000; Vilà *et al.* 1999; Scandura *et al.* 2001). In the Old World, mtDNA data suggests that most populations are genetically differentiated with the exception of neighbouring populations such as those in Spain and Portugal, or recently invaded areas such as France, where Italian wolves have migrated (Taberlet *et al.* 1996; Vilà *et al.* 1999; Randi *et al.* 2000). In Western Europe, genetic subdivision may reflect recent habitat fragmentation that occurred over the past few hundred years with the loss of forests and, more importantly, a dramatic decrease in the size of all wolf populations due to human persecution (Wayne *et al.* 1992; Vilà *et al.* 1999). Genetic subdivision in Asian populations is not well known, however, recent mitochondrial DNA studies have found two subspecies of highly distinct Himalayan and lowland Indian wolves (*C. l. pallipes* and *C. l. chanco*, respectively) (Aggarwal *et al.* 2003; Sharma *et al.* 2004).

The presence of genetic subdivision in Europe contrasts with the patterns in North America where clinal variation in microsatellite alleles may exist over short distances (Forbes and Boyd 1996, 1997) although it is less apparent at a continental scale (Roy *et al.* 1994b). Similarly, mtDNA haplotypes are shared across large distances (Wayne *et al.* 1992; Vilà *et al.* 1999) but some geographic patterns also are evident. For example, mitochondrial RFLP haplotype W3 was common in Alaska and Northwest Territories but absent from populations in eastern Canada (Wayne *et al.* 1992). Conversely, RFLP haplotype W1 was absent in

Alaskan wolves but common in eastern Canada. A similar pattern was observed for mitochondrial control region sequences (Vilà *et al.* 1999). Conceivably, these weak clinal patterns reflect prior Pleistocene isolation in southern and Alaskan refugia followed by expansion and mixing during interglacials. Water barriers and differences in prey may also result in differentiation. For example, a recent study found that wolves specialising on different caribou herds in the Canadian north-west as well as populations on Banks and Victoria Islands were differentiated (Carmichael *et al.* 2001). Finally, another level of complexity is suggested by the recent finding that the Great Lakes wolf population may have been a distinct red wolf-like canid, *Canis lycaon*, which is now interbreeding with grey wolves that have migrated into eastern Canada after the last glaciation and coyotes which have entered the region in the past 100 years (see below; Wilson *et al.* 2000). Regardless, North American grey wolves proved not to be as dramatically structured and reduced in variation as their Old World counterparts as evidenced by the observation that population variability was high and levels of differentiation were low (Wayne *et al.* 1992; Roy *et al.* 1994b, 1996; Vilà *et al.* 1999).

The most highly differentiated North American grey wolf population is the Mexican wolf. Except for a reintroduced experimental population, this subspecies was thought to be extinct in the wild and exists only in three captive populations, each initiated by a small number of founders (García-Moreno *et al.* 1996; Hedrick *et al.* 1997). Two of the captive Mexican wolf populations displayed a

single divergent mtDNA haplotype found nowhere else that was more closely related to a subset of Old World haplotypes than to any New World haplotype. This suggested that Mexican wolves shared a more recent ancestry with wolves from the Old World and the basal position of the Mexican wolf sequences in phylogenetic trees and analysis of historic museum specimens suggested that the Mexican wolf was a relict form stemming from an early invasion of grey wolves from Asia (Wayne *et al.* 1992; Vilà *et al.* 1999; J. A. Leonard *et al.* unpubl.).

The red wolf and Algonquin wolf

Interbreeding between highly mobile species, such as wolves and coyotes, may result in the development of large hybrid zones. The grey wolf once ranged throughout most of North America and parts of Mexico, but over the past few hundred years, wolves have been eliminated from the USA and Mexico. Similarly, the red wolf was exterminated by about 1975 from throughout its historic distribution which included much of the south-eastern USA, although it has since been reintroduced to a refuge in North Carolina (Parker 1987). Coyotes interbred extensively with red wolves as they approached extinction (Nowak 1979) and, consequently, mtDNA haplotypes and microsatellite alleles otherwise unique to coyotes are found in red wolves (Wayne and Jenks 1991; Roy *et al.* 1994b). However, an extensive genetic analysis characterising microsatellite and mtDNA variation in coyotes, grey wolves and historic and recent red wolves found no markers unique to red wolves. Instead only haplotypes and microsatellite alleles identical or very



Adult male red wolf with pup.
Museum of Life and Sciences,
Durham, USA. 2002.

Greg Koch

similar to those in grey wolves and coyotes were found (Roy *et al.* 1994a, 1994b, 1996). Consequently, an origin of the red wolf through hybridisation of grey wolves and coyotes in historic times or earlier was postulated (Wayne and Jenks 1991; Roy *et al.* 1994b, 1996; Reich *et al.* 1999). Similarly, genetic evidence for hybridisation between grey wolves and coyotes from Minnesota and eastern Canada suggested a hybrid form similar to that which may have existed for the red wolf (Lehman *et al.* 1991; Roy *et al.* 1994b, 1996). Thus, the genetic data imply both that significant hybridisation has occurred between the two species and that introgression of coyote genes into the wolf population has occurred over a broad geographic region.

However, new genetic results question these conclusions (Wilson *et al.* 2000). Detailed genetic analysis of eastern Canadian wolf-like canids and coyotes has found divergent mtDNA control region haplotypes with a distribution centred at Algonquin Provincial Park, Ontario. These divergent haplotypes appear to be phylogenetically similar to those of red wolves, which in turn are grouped with haplotypes of coyotes. These results may indicate that the smallish grey wolf that formerly inhabited the Great Lakes areas and the red wolf, are the same species, designated as *Canis lycaon* (Wilson *et al.* 2000). These authors suggest that the Algonquin wolf is a native New World wolf-like form that evolved independently from North American coyote-like ancestors (see Nowak 2002 for an alternative view). Finally, a recent analysis of coyotes in the south-eastern USA has shown that one dog haplotype appeared in multiple individuals across a large area (Adams *et al.* 2003). This suggests an ancient coyote-domestic dog hybridisation event when the first coyotes were expanding into eastern habitats formerly occupied by red wolves.

Wolf-dog hybridisation

In the wild, hybridisation between grey wolves and dogs is likely to be most frequent near human settlements where wolf density is low and habitats are fragmented, and where feral and domestic dogs are common (Boitani 1983; Bibikov 1988). The genetic integrity of wild wolf populations has been a concern among some conservationists (Boitani 1984; Blanco *et al.* 1992; Butler 1994), although the majority of wolf populations show no evidence of hybridisation (Vilà and Wayne 1999). However, genetic studies have detected limited wolf-dog hybridisation in Bulgaria, Italy, Latvia, Scandinavia and Spain (Dolf *et al.* 2000; Randi *et al.* 2000; Randi and Lucchini 2002; Andersone *et al.* 2002; Vilà *et al.* 2003a; Vilà *et al.* unpubl.).

Conservation implications

Several conservation implications are suggested by the genetic results. First, because the Mexican grey wolf is genetically and physically distinct, and historically isolated from other grey wolves (Nowak 1979), the breeding of pure Mexican wolves in captivity for reintroduction into

the wild is advised. Second, because most wolf populations in North America are not strongly differentiated genetically, and gene flow is high among populations, reintroduction need not include only the nearest extant populations as source material. However, the reintroduction of wolves from populations where hybridisation with coyotes has occurred is perhaps not advisable (see below). Finally, genetic analysis of recolonised populations in Montana and France has found that high levels of genetic variation can be preserved (Forbes and Boyd 1997; Scandura *et al.* 2001).

The grey wolf has been divided into as many as 32 subspecies worldwide (Hall and Kelson 1959). Nowak (1995) suggested that the 24 North American subspecies should be reduced to five. However, rates of gene flow among North American wolf populations are high, and differentiation by distance characterises the genetic variation of wolves at some geographic scales. In this sense, typological species concepts may be inappropriate because geographic variation in the wolf is distributed along a continuum rather than being partitioned into discrete geographic areas delineated by fixed boundaries. A focus on locality-specific adaptations to prey size or climate (e.g., Thurber and Peterson 1991; Carmicheal *et al.* 2001) or size variation with latitude may be a more appropriate guide to conservation rather than arbitrary boundaries of a continuously distributed and high mobile species (Crandall *et al.* 2000). However, the discovery of genetically distinct populations of Indian wolf suggests they should be the focus of immediate research and the object of conservation concern. Finally, although contemporary wolf populations in Europe appear more genetically subdivided than their North American counterparts (Wayne *et al.* 1992; Vilà *et al.* 1999; Randi *et al.* 2000), the North American pattern might well reflect the ancestral condition in western Europe prior to habitat fragmentation and population decimation. Therefore, efforts to increase gene flow among European wolf populations to levels similar to that in North America could be defended.

The possible presence of a hybrid zone between a native north-eastern wolf species, and coyotes and grey wolves (see above) complicates taxonomic and conservation recommendations. If *C. lycaon* is a distinct species, conspecific with the red wolf, then captive breeding and conservation efforts *in situ* may be urgently needed. If *C. lycaon* is a hybrid between grey wolves and coyotes that is due to human-induced habitat changes and predator control efforts, then further conservation efforts may not be warranted (Jenks and Wayne 1992; Wayne and Brown 2001). For the hybridisation process to be of conservation concern, even hybridisation between a unique North American wolf and other canids, it should be caused by human activities rather than natural processes, such as glacial induced range expansions. Additional genetic data

involving multiple mitochondrial, nuclear, and Y-chromosome markers are needed to better test alternative hypotheses for the origin of the red wolf and the Algonquin wolf. Finally, wolf-dog hybridisation is a non-natural occurrence that fortunately may be of concern only in a few European populations (see above).

Perhaps of greater concern is the loss of genetic variation in isolated wolf populations in the Old World (see above). Inbreeding depression has been documented in captivity (Laikre and Ryman 1991; Laikre *et al.* 1993; Federoff and Nowak 1998; Fredrickson and Hedrick 2002; but see Kalinowski *et al.* 1999). Italian, Scandinavian and Isle Royale wolves have levels of average relatedness approaching inbred captive populations (see above), and could conceivably suffer a decrease in fitness that would eventually affect population persistence (Wayne *et al.* 1991a; Mace *et al.* 1996; Hedrick and Kalinowski 2000). High levels of gene flow likely characterised Old World populations in the past, so there is reason to restore past levels of gene flow in parts of Europe, either through habitat restoration and protection along dispersal corridors or through translocation. Future research should be aimed at monitoring and predicting genetic changes that will occur in wolf populations and trying to determine any possible population effects.

11.2.4 Kit fox and swift fox

Small canids such as foxes may have limited dispersal ability and be less able to traverse topographic barriers. Moreover, due to shorter dispersal distances, small canids may show a more pronounced pattern of genetic differentiation with distance and population subdivision. The small arid land foxes of North America are habitat specialists and relatively poor dispersers. In California, for example, the kit fox of the San Joaquin Valley, whose range is circumscribed by the coastal mountain range to the west and the Sierra Nevada mountain range to the east, is considered a distinct subspecies (*Vulpes macrotis mutica*) and is protected by the U.S. Endangered Species Act (Hall 1981; O'Farrell 1987). Populations to the east of the Rocky Mountains are collectively referred to as swift foxes (*V. velox*), and those to the west as kit foxes (*V. macrotis*). However, the two forms hybridise in north central Texas and are recognised as conspecific by some authors (Packard and Bowers 1970; Rohwer and Kilgore 1973; Nowak and Paradiso 1983; O'Farrell 1987; Dragoo *et al.* 1990).

Population variation and differentiation

Mitochondrial DNA analyses suggests that genetic divergence is related to the distance between populations and the severity of the topographic barriers separating them (Mercure *et al.* 1993). A major genetic subdivision within the kit-swift fox complex distinguished populations

from the east and west side of the Rocky Mountains, consistent with the taxonomic distinction between *V. macrotis* and *V. velox*. The divergence between these taxa was nearly as great as that between them and the Arctic fox (*Alopex lagopus*), classified in a separate genus. Furthermore, within each of the two major kit-swift fox mtDNA clades, genetic distances among populations tended to increase with geographic distance (Mercure *et al.* 1993). The distinct phylogeographic pattern in kit-swift foxes contrasts with the lack of pattern observed in coyotes and grey wolves (Lehman and Wayne 1991) and suggests that the kit and swift fox may be two distinct species. However, the two forms hybridise in a contact zone in New Mexico, and microsatellite evidence indicates hybridisation occurs freely within the hybrid zone (Dragoo and Wayne 2003; J. W. Dragoo unpubl.). Finally, mitochondrial DNA data support subspecific distinction of the San Joaquin and Mexican (*V. m. zinseri*) kit foxes (Mercure *et al.* 1993; Maldonado *et al.* 1997). The latter appears to have reduced variation relative to USA populations.

Conservation implications

The San Joaquin and Mexican kit fox are genetically distinct populations that are related to kit foxes west of the Rocky Mountains. This degree of distinction suggests a limited history of isolation and provides some support for special preservation efforts. Further, the small population size and isolation of the Mexican form and absence of conservation efforts raise concern for its persistence. Topographic barriers, such as the Colorado River, or habitat barriers appear to influence geographic differentiation, but the predominant pattern within clades is one of geographic differentiation with distance. The scale of differentiation with distance is much finer in kit-swift foxes than in large canids reflecting differences in dispersal abilities and suggesting that a larger number of genetic units of conservation concern can be defined in small canids. With respect to the design of reintroduction programmes, source stocks for small canids should in general be drawn from smaller geographic areas than the large canids. For example, given the mtDNA findings, the recent use of foxes from Colorado and South Dakota rather than New Mexico or Texas as a source for a reintroduction into Canada's Saskatchewan Province, appears to have been appropriate (Scott-Brown *et al.* 1987).

11.2.5 Island fox

The island fox (*Urocyon littoralis*) is a Critically Endangered species found only on the six Channel Islands off the coast of southern California (Gilbert *et al.* 1990; Wayne *et al.* 1991b). The island fox is an insular dwarf, about two-thirds the size of its mainland ancestor, the

gray fox (*U. cinereoargenteus*) (Collins 1991a; Wayne *et al.* 1991b). As suggested by the fossil and geologic record, about 16,000 years ago the three northern islands, which at that time were connected to one another, were colonised by foxes from the mainland. As sea level rose, 9,500 to 11,500 years ago, the northern islands were separated. About 4,000 years ago, foxes first arrived on the southern Channel Islands and were probably brought there by Native Americans. Consequently, genetic variation in the island populations should correlate with island area and founding time (Wayne *et al.* 1991b).

Population variation and differentiation

In general, predictions about genetic variation, island area and colonisation time were supported by molecular genetic analyses (Gilbert *et al.* 1990; Wayne *et al.* 1991b; Goldstein *et al.* 1999). The small, late colonised, San Nicolas population was invariant in all genetic markers surveyed including multi-locus DNA fingerprints and 19 microsatellite loci (Gilbert *et al.* 1990; Goldstein *et al.* 1999; Roemer *et al.* 2001a, 2002). Only inbred mice strains show a similar lack of variation, but no other wild population except the inbred, eusocial naked mole rats (*Heterocephalus glaber*) approaches this level of monomorphism (Faulkes *et al.* 1997). Similarly, the smallest island, San Miguel, had low levels of variation. In contrast, the large islands, Santa Catalina, Santa Rosa and Santa Cruz, had higher levels of variation. However, the Santa Cruz Island population, although it was founded early, had lower levels of variation than expected and Santa Catalina, although founded last, had the highest levels of variation. Finally, mitochondrial DNA data suggested that Santa Catalina Island may have been colonised multiple times from southern and northern islands.

All populations were well differentiated (Wayne *et al.* 1991b; Goldstein *et al.* 1999). The island foxes did not share mitochondrial DNA sequences with the mainland gray fox and some populations had unique haplotypes. For example, within the southern group of islands, the small population on San Nicolas possessed a unique mtDNA haplotype. Similarly, island populations had unique multilocus fingerprint bands and microsatellite alleles and differed in allele frequencies. Consequently, foxes could be correctly classified to island of origin, and haplotype trees resolved an evolutionary history of colonisation consistent with the archaeological record (Wayne *et al.* 1991b; Goldstein *et al.* 1999).

Conservation implications

Fox populations on five of six islands have decreased dramatically over the past 10 years (Roemer *et al.* 2002). On the three northern islands, the decline was due to predation by golden eagles (*Aquila chrysaetos*) (Roemer *et al.* 2001b, 2002). On Santa Catalina Island, the decline was

due to a distemper epidemic and on San Clemente, a more gradual decline likely reflected predator control efforts of the loggerhead shrike (*Lanius ludovicianus migrans*) reintroduction programme (Roemer and Wayne 2003). In each case, genetic management of the remaining population is needed. On the northern islands, captive breeding is necessary to restore the wild populations, and preliminary studies have suggested that the captive population has sampled a limited subset of variation in the wild implying additional founders would be a beneficial addition to the captive breeding programme (Aguliar *et al.* unpubl.; M. Gray *et al.* unpubl.). On San Miguel Island, the wild population may be extinct, but on the other two northern islands, a dynamic exchange of wild and captive born foxes to enrich genetic variability is conceivable. On Santa Catalina Island, foxes have disappeared from about 90% of the island and a captive breeding programme of survivors has been established to assist in replenishing the loss of populations. However, a wild reservoir of over 150 foxes exists on the far western end of the island and genetic data indicate they provide a more genetically variable source for reintroduction (Aguliar *et al.* unpubl.). On San Clemente Island, several hundred individuals remain in the wild and significant genetic loss is unlikely to have occurred. There should be immediate efforts to stabilise the population and prevent further decline (Roemer and Wayne 2003).

The genetic results suggest that each island population should be treated as a separate conservation unit. Further, low levels of genetic variation in each island population relative to mainland gray foxes imply that they may be more vulnerable to environmental changes (Frankham *et al.* 2002). In previous conservation plans, the species has been treated as a single taxonomic unit with a combined population of about 8,000 individuals (California Code of Regulations 1992). However, as is now clear, by virtue of their isolation and small size, the islands are more vulnerable than an equivalently sized mainland population. Each island should be designated an independent unit with regard to conservation and at least five populations should be considered in immediate danger of extinction (Roemer and Wayne 2003). Study of captive populations combined with careful genetic management may allow successful reintroduction and more informative management of wild populations in the future.

11.2.6 Darwin's fox

On Chiloé Island, off the west coast of Chile, Charles Darwin observed and was the first to describe a small endemic fox, *Pseudalopex fulvipes*. Darwin's fox is a unique species found principally on Chiloé Island and has the smallest geographic range of any living canid (Osgood 1943; Cabrera 1958). There are perhaps less than 500 foxes currently in existence and none in captivity. Darwin's

fox was thought to be recently isolated from mainland foxes given that the channel separating Chiloé from the continent is only about five kilometres wide, and the island was likely connected to South America when sea levels were lower during the last glaciation (*c.* 13,000 years B.P., Yahnke *et al.* 1996). However, the recent discovery of Darwin's fox on the mainland in central Chile, in Nahuelbuta National Park, 350km from Chiloe (Medel *et al.* 1990), and where they are sympatric with the mainland chilla (*P. griseus*), suggested that Darwin's fox may be a distinct species.

Genetic variation and differentiation

Phylogenetic analysis of mitochondrial DNA sequences confirmed the distinct species status of Darwin's fox and showed the island and single mainland populations are conspecific. Only three haplotypes were found, and only one of the mainland implying low genetic variation. Further, modest levels of genetic distinction between mainland and island populations supports their designation as a distinct subspecies. The genetic results show that Darwin's foxes diverged early in the radiation of Chilean foxes, and are at least as divergent from the chillas and culpeos (*P. culpaeus*) as the latter two are from one another. These results indicate that Darwin's fox is a relict form, having evolved from the first immigrant foxes to Chile after the land bridge formed between North and South America about 2–3 million years ago (Webb 1985; Yanke *et al.* 1996). Finally, a recent survey of other potential mainland localities where Darwin's fox may have formerly lived has found at least one locality approximately 150km north of Chiloé Island where foxes existed historically (C. Vilà *et al.* unpubl.).

Conservation implications

Darwin's fox is genetically distinct and appears to be the progenitor of the mainland fox species. The genetic results suggest it had a previous distribution on the mainland, rather than having been introduced there by humans. Darwin's fox also has a morphology unlike mainland foxes, and occupies a restricted and unique temperate rainforest habitat. Darwin's fox needs to be considered a distinct species of urgent conservation importance. The island population needs greater protection and captive populations need to be established. Captive breeding and observation of Darwin's foxes might provide better understanding of the biology and status of this Critically Endangered species that will assist in its conservation. The recent extinction and decline of four island fox populations of similar size to that of the Darwin's fox highlight its vulnerability. Finally, the presence of Darwin's foxes on the mainland currently, and in recent history, provides a

mandate for introduction, possibly using the island population as a source (C. Vilà *et al.* unpubl.).

11.3 Conclusions

In general, the smaller fox-like canids show higher levels of variation between and within populations. These differences reflect higher densities and low levels of mobility in small canids. Insular canids, such as the island fox, Darwin's fox and the Isle Royale wolf, have the lowest levels of genetic variation but high levels of differentiation from mainland populations. A similar pattern is evident when habitats have been subdivided and populations isolated by human activities. For example, Scandinavian and Italian wolves have low levels of variation within populations but high levels of differentiation reflecting a recent history of isolation and population bottlenecks (Randi *et al.* 2000; Vilà *et al.* 2003b). The Ethiopian wolf, arguably the most threatened canid (despite its downgraded status in this plan), has the lowest levels of variation of any studied canid. In contrast, African wild dog genetic patterns appear dominated by ancient vicariance events such as Pleistocene isolation of southern and eastern populations followed by intermixing. However, populations in Kruger National Park and Kenya may have recently lost genetic variation due to population bottlenecks. Finally, interspecific hybridisation may occur in disturbed populations especially if one species is rare and the other abundant and the rate of encounters is high due to the presence of concentrated resources such as refuse dumps. Hybridisation with domestic dogs may threaten preservation of the unique genetic characteristics of the largest remaining population of the endangered Ethiopian wolf, but does not appear as a consequential threat to grey wolves.

Molecular genetic analysis supports species distinction for Darwin's fox, kit and swift foxes and the island fox. Analysis of populations within species have uncovered important genetic and phenotypic units including each of the island populations of island fox, the San Joaquin and Mexican kit fox, the Mexican wolf, two subspecies of Indian wolf and the Algonquin wolf, West and South African wild dogs, and New and Old World wolves. These distinct conservation units warrant separate breeding and in situ management. Interbreeding should be avoided in the absence of evidence for inbreeding depression (e.g., Hedrick and Kalinowski 2000). The next phase in genetic research on canids should focus on study of both neutral and fitness related genes so that both history and population adaptation can be assessed. This information will be valuable to conservation programmes (Crandall *et al.* 2000).

Assessing and Managing Infectious Disease Threats to Canids

M. K. Laurenson, S. Cleaveland, M. Artois and R. Woodroffe

12.1 Introduction

Infectious diseases in wildlife have attracted increased attention in recent years and can be an important and intractable extinction risk for many species. A number of reviews have highlighted how disease issues are particularly relevant to canids whether because canids are the source of human diseases such as rabies, leishmaniasis or hydatid cysts (*Echinococcus granulosus*), or because threatened canids have suffered high disease-related mortality (Young 1994; Funk *et al.* 2001; Cleaveland *et al.* 2002; Woodroffe

et al. 2004). Examples of disease in threatened canid populations are summarised in Table 12.1, which illustrates that rabies and canine distemper (CDV) are currently of greatest concern (Macdonald 1983; Appel *et al.* 1995; Deem *et al.* 2000; Woodroffe *et al.* 2004). Other diseases such as parvovirus, anthrax or otodectic mange have caused more sporadic outbreaks or may have affected populations in less obvious ways (Creel *et al.* 1995; Steinel *et al.* 2001).

This apparent association between canids and disease may arise in several ways. First, some canid species, most specifically foxes, coyotes and jackals, are opportunistic

Table 12.1. Local extinctions and crashes of free-ranging canid populations known to have been caused by infectious disease (based on Woodroffe *et al.* 2003).

Species	Pathogen	Effect	Reference
African wild dog	Rabies	Local extirpation of population of 50–70 dogs	Gascoyne <i>et al.</i> 1993 ; Kat <i>et al.</i> 1995
African wild dog	Rabies	Pack extirpated, all four dogs died	Scheepers and Venzke 1995
African wild dog	Rabies	75% of 12 dogs	Hofmeyer <i>et al.</i> 2000
African wild dog	Rabies	50% of 10 packs	J.W McNutt, pers. comm.
African wild dog	CDV	49/52 dogs died	Van de Bildt <i>et al.</i> 2002
African wild dog	CDV	All 12 dogs in pack died	Alexander <i>et al.</i> 1996
African wild dog	Anthrax	unknown, but 4/5 dogs confirmed <i>B. anthracis</i> 4/8 affected pups died, 3/3 adults survived	Turnbull <i>et al.</i> 1991 Creel <i>et al.</i> 1995
Ethiopian wolf	Rabies	53/76 known wolves died	Sillero-Zubiri <i>et al.</i> 1996b
Ethiopian wolf	CDV	Outbreak in sympatric domestic dogs. Effect on wolf population not monitored but population slow to recover from rabies outbreak	Laurenson <i>et al.</i> 1998
Ethiopian wolf	Rabies	~75% died/missing from subpopulation of 80 wolves	Randal <i>et al.</i> submitted
Kit fox	Rabies	96% decline over four years	White <i>et al.</i> 2000
Island fox	CDV	89% of 1,340 foxes disappeared	Timm <i>et al.</i> 2000
Island fox	Heartworm	78% seroprevalence, potentially some mortality in older individuals	Crooks <i>et al.</i> 2001
Blanford's fox	Rabies	75% of four foxes died	Macdonald 1993
Mednyi Arctic fox	Otodectic mange		Goltsman <i>et al.</i> 1996
Grey wolf	CPV	Pups died: Affected recruitment in one area, although overall population not affected	Mech and Goyal 1993, 1995
Grey wolf, coyote	Sarcoptic mange	Alberta	Todd <i>et al.</i> 1981a
Red fox	Sarcoptic mange	Various countries have caused population crash/limitation	Morner 1992; Danell and Hornfeldt 1987; Lindstrom <i>et al.</i> 1994; Tsukada <i>et al.</i> 1999; Harris and Baker 2001

generalists that can flourish in human-altered landscapes. Thus, their populations may be of sufficient density and size to allow some pathogens to persist. Second, the ecology of canids may expose them to infection. They may eat infected prey (Brand *et al.* 1995; Creel *et al.* 1995; Scheepers and Venzke 1995), and they frequently live in closely-knit groups that are often intolerant of other conspecifics. Interactions with conspecifics, whether social or aggressive may therefore be frequent (Woodroffe *et al.* 1997; Haydon *et al.* 2002). All these factors facilitate the transmission of infection to-and-between individuals. Third, most wild canids are closely related to the domestic dog and, therefore, susceptibility to most pathogens is shared. Where these dogs are not routinely vaccinated against common canid pathogens there is a potentially large population that can be a pathogen reservoir. Contact between wild canids and domestic dogs may be frequent in inhabited areas where wild canids coexist with humans, where domestic dogs invade protected areas, or where wide-ranging canids leave protected areas (Woodroffe and Ginsberg 1998; Woodroffe *et al.* 2004). As free-ranging domestic dogs and wild canids may directly compete for similar communal resources, such as carcasses or human refuse, contact may be more frequent than, for example, between wild and domestic felids.

Given this context, we aim to provide essential background information on disease epidemiology for wild canid managers in this chapter, but do not provide detailed descriptions of diseases or theory that can be obtained in other texts. We also aim to provide some practical information for assessing and managing disease threats, as a starting place for the field manager contemplating this issue.

12.2 Ecology of disease

An understanding of disease dynamics in host populations is fundamental to the management of canid disease. As space here is insufficient, we refer readers to a number of reviews to provide this essential background, including Anderson and May (1991), Hudson *et al.* (2002), and Woodroffe *et al.* (2004). However, it is necessary to highlight two important points. First, all the pathogens that have caused a problem in small or threatened populations of canids have the ability to infect a wide range of species and are generalists (Cleaveland *et al.* 2002). Indeed, epidemiological theory predicts that pathogens that cause major host mortality or reduce fertility are unlikely to be able to persist in small populations (Lyles and Dobson 1993). Second, these generalist pathogens must, therefore, persist in another reservoir population (Haydon *et al.* 2002). From there, they can spill-over and cause one-off or repeated epidemics in the target, threatened population.



Wildlife Rescue Association of British Columbia

Captured coyote with mange at animal rescue facility. The disease is common in coyotes in the area, and can be a symptom of a weakened immune system or other illness. Burnaby, British Columbia, Canada, 2003.

12.2.1 What is the disease reservoir?

Identifying the reservoir host for the disease is often crucial when considering the management of disease threats to canids, as it determines the options available and their likelihood of success. Identifying the reservoir is rarely a trivial task: the ultimate test of the identity of the reservoir is in some way to remove the pathogen agent from the putative reservoir community and then monitor incidence of disease in other target species in the community (Haydon *et al.* 2002). Unfortunately, this is often difficult if not impossible to carry out for reasons of scale, logistics or, for example, when the environment is the reservoir (as for Anthrax *B. anthracis*). However, other strands of evidence can be pieced together which, although not providing unequivocal proof, will suffice in most situations where practical decisions have to be made. First, epidemiological evidence in the form of case-control and cohort studies (Thrusfield 1995) may identify risk factors, such as contact with a particular species. Second, evidence of infection in putative host species, through antibody detection (serology) or the isolation of the infectious agent is a useful step, but must be interpreted with caution: not all natural hosts are reservoir hosts and may be 'spill-over' hosts. However, the same genetic or antigenic character of the pathogens from target and putative reservoir species should be identified. Note also that the level of incidence or seroprevalence in a one-off cross sectional study does not give any additional evidence. Pathogens may persist in reservoirs at high or low prevalence: the critical issue is persistence and this can only be determined through longitudinal studies (Haydon *et al.* 2002).

The major pathogens of concern for canids can infect a wide spectrum of carnivores and even other mammal orders. Different host species may be reservoirs in different

ecological and epidemiological situations. Thus, it is often inappropriate to extrapolate from one situation to another. Due consideration must be given to the carnivore community of interest, their relative density and the likelihood of intra- and interspecies contact.

12.3 Assessing the type and severity of disease threat

Assessing the presence and severity of a disease threat, before potential threats become a reality, is both a key issue for managers of wild canids and another thorny problem (Wobeser 1994). In some situations, the potential for a disease outbreak was identified in general terms before epidemics struck (e.g., Gottelli and Sillero-Zubiri 1990), but in many others, an epidemic was unforeseen (Gascoyne *et al.* 1993). In only a few cases, had management actions been put in place before the epidemic struck. Unfortunately, in many cases the management actions, for one unforeseen reason or another, did little to ameliorate the situation (see Woodroffe *et al.* 2004).

Clearly, the severity of the disease threat depends on a number of factors: first, the incidence of each potential disease in the reservoir; second, the potential impact of the disease on target individuals and the population; and third, the probability of that disease being transmitted from the reservoir to the target species. Assessing and quantifying all these aspects is a considerable challenge. The potential methods of assessment and surveillance will depend on the disease in question, the type of reservoir (if known) and the ecology of the system.

12.3.1 Rabies

Rabies is probably the disease of the greatest concern to managers of wild canids. Rabies has an impact on public health and livestock economies and thus data on its presence and incidence is invariably of better quality than that for other pathogens. However, in many areas, particularly as distance to the laboratory increases and veterinary and transport infrastructure declines, samples from suspected rabies cases are not collected by veterinary services and sent for laboratory diagnosis. For example, in Ethiopia, although rabies is found in a large proportion of the country, laboratory diagnosed rabies cases are almost always confined to the capital city and nearby areas.

Hospital records are another source of information in rabies prevalent areas. People bitten by suspected rabid animals will often seek treatment at clinics and hospitals and thus data on the incidence of dog bites and human rabies cases can be valuable (Cleaveland *et al.* 2002). However, hospital records will invariably underestimate the severity of the problem, as many people may not seek

treatment, particularly where traditional medical practices are strong (Cleaveland *et al.* 2002).

However, where rabies is prevalent, the knowledge of the presence of the disease and its clinical signs are relatively well known in local communities. For example, in Tanzania, 80% of samples submitted by local people from suspected rabies cases were positive on laboratory analysis and thus more accurate than diagnostic tests for many diseases such as tuberculosis (Cleaveland *et al.* 2003). Thus, reports of suspected rabies cases can yield valuable data on occurrence and incidence if questionnaires are done in a systematic way and detailed information on suspected rabies cases is obtained.

12.3.2 Other diseases

The presence of other diseases can be more difficult to assess, as clinical signs are less well recognised and laboratory diagnoses are less likely to have been obtained. This is particularly true for endemic diseases, those with low mortality rates, or those affecting fertility or recruitment rather than causing mortality. When a major epidemic occurs, however, the probability of it being identified as a separate disease with accurate descriptions of clinical signs by local informants is increased (e.g., CDV, Laurenson *et al.* 1998). Screening of blood samples from putative reservoir hosts for antibodies to potential pathogens (indicating previous infection by the pathogen) can be instructive for microparasites. This approach has some drawbacks and will be less successful if the disease occurs in a more epidemic pattern in the reservoir hosts or if mortality rates are high, leaving few survivors with serological evidence of infection. In addition, antibodies to some pathogens have a short half-life (e.g., rabies) and thus will decline with time, leaving no serological evidence. Examination of faecal samples or post-mortem samples can be useful to determine the occurrence of macroparasite infection (see Box 12.1).

12.3.3 Type of reservoir

When domestic carnivores, principally dogs, are the reservoir, the presence of pathogens circulating in the reservoir population disease will be easier to identify. Surveillance for all diseases is obviously more straightforward in a domestic host, as will be obtaining blood samples for serological screening. Obtaining information on the occurrence and incidence of disease in wildlife reservoirs is more problematic, not least because post-mortem samples are difficult to obtain as carcasses disappear and decompose quickly (Gulland 1995). However, reports from local communities, wildlife officers, researchers, farmers, and hunters may provide valuable information. Maximum advantage should be taken of any carcasses found (see Box 12.1 for guidance on necropsy

techniques). Even if carcasses are relatively decomposed, appropriate samples should still be taken, as modern techniques such as PCR are extremely powerful (Table 12.2). Serum can also be obtained from frozen lung or muscle tissues after freezing. When major epidemics in wildlife occur, there is clearly an improved chance of carcass detection and collecting fresh samples.

Where wild species are the reservoir, obtaining blood samples for serological analysis requires trapping and handling animals and is clearly difficult. Thus, when handling occurs for other reasons, such as fitting radio-collars, it is imperative that these rare opportunities are not wasted: blood samples should always be taken and

serum stored for future analyses. Similarly, samples may be obtained from species that are hunted for trophies, or harvested or culled as problem animals. Results of laboratory analyses, even if not published, should be willingly shared with management and other interested parties.

12.3.4 Pathogen and host ecology

The ecology of both the pathogen and host population are also important in determining the severity of threat. From a pathogen perspective, aside from the fundamental issue of pathogen pathogenicity, both transmission mode and

Table 12.2. Collection and storage of carnivore post-mortem samples for disease diagnosis. Note that CITES permits from both exporting and importing countries are required for shipping samples from most canids internationally. Tissue samples should be treated gently and those to be stored in formalin should be 5–10mm thick and placed in about 10 times their volume of 10% buffered formalin. Hollow organs should be opened and their contents (e.g., faeces) removed before fixation. Once the tissue is fixed, one can drain away most of the formalin, leaving just enough to keep them moist, and submit them for examination by post if local experts are unavailable. It is always wise to split samples into at least two (but preferably more) sub-samples or aliquots which are separately stored and transported. Catastrophic equipment failures or losses in transit are not uncommon.

Tissue	Storage	Temperature	Test
Brain Salivary Gland	Straw sample (glycerol saline)	-20°C (but can be stored at ambient temperature for several weeks)	Rabies (FAT, Virus isolation)
Brain Salivary Gland	Plain (cryotubes)	Liquid nitrogen (-20°C if liquid nitrogen not available, but CDV not well preserved)	Rabies (FAT, virus isolation) CDV (virus isolation from brain)
Brain	10% buffered formalin	Ambient temperature (do NOT freeze)	Histopathology – rabies, CDV and other infections
Lymph nodes (all)	Plain (cryotubes)	Liquid nitrogen -20°C	CDV isolation, particularly bronchial and pulmonary nodes
Lymph nodes (all)	10% buffered formalin	Ambient temperature (do NOT freeze)	Histopathology
Faecal samples	Plain	Freeze -20°C	CPV virus/antigen detection
Faecal samples	High-grade ethanol	Ambient temperature	Molecular genetic analyses (population genetic), PCR diagnostics
Faecal samples	5% formalin	Ambient temperature	Hydatid coproantigen assays
All tissues – including brain, salivary gland, lymph nodes, lung, liver, kidney, spleen, small intestine	10% formalin	Ambient temperature (do NOT freeze)	Histopathology – all diseases
All tissues – including brain, salivary gland, lymph nodes, lung, liver, kidney, spleen, small intestine	High-grade ethanol	Ambient temperature	PCR diagnosis – rabies, CDV, CPV, CAV, hydatid, other pathogens
Blood smear	Microscope slide – air-dried	Ambient temperature	Haemoparasites, anthrax
Blood, exudates, tissue fluids or homogenates	FTA filter-paper cards*	Ambient temperature (keep DRY)	PCR diagnosis
Blood	Tris Buffer – ‘Easy Blood’**	+4°C	PCR diagnosis of haemoparasites

*FTA® Classic Card – Whatman Bioscience

Place drop of fluid onto an FTA-treated card, allow to dry and place card into sealed pouch with drying sachet. The FTA card lyses all cells and inactivates bacteria and viruses (rendering samples non-hazardous), but allows clean DNA to be retained and easily transferred to PCR reaction vessels for performing amplifications. Samples on cards can be stored for up to 10 years at room temperature.

** ‘Easy Blood’

Box 12.1. Post-mortem examination of canids (adapted from Woodroffe *et al.* 1997).

Post-mortem examinations are best carried out by vets or other qualified personnel but as the collection of fresh tissue samples as soon after death as possible is of critical importance in obtaining a diagnosis, fieldworkers should carry these out immediately if these personnel are not available. Photographs or video footage should be taken throughout the examination. Observers doing necropsies should wear protective clothing, particularly gloves, and disinfectant measures taken on completion. It is often possible to have samples examined by local veterinary laboratories, but ensure duplicate samples are taken (see Table 12.2). (Linda Munson at UC Davis may be available to give advice on sample analysis – lmunson@green.ucdavis.edu).

Equipment that should be held for post-mortem examination and sample collection

Post mortem: Strong, sharp knives, a sharpening steel or stone, scissors, forceps, scalpel handles and blades, a hacksaw or rib cutters, and possibly a small hatchet.

Sample collection: Formalin (10%), glycerol saline, drinking straws (5mm diameter), various sized leakproof containers, sterile swabs.

Carrying out the post-mortem

1. Begin with a visual examination of the animal, and then palpate any abnormalities. Record the nutritional state (body condition) of the animal.
2. Cut into the right axilla (armpit) and coxofemoral (hip) joint, and turn back both right legs. Then make a shallow incision along the ventral midline, cutting through the skin from the chin to the pelvis. Do not cut across hair: instead, roll the skin back after making the first incision, and cut underneath, which preserves the edge of the knife. Peel the skin back from the underside of the dog.
3. Open the abdomen cavity by carefully cutting through the abdominal wall from the xiphoid cartilage along the last rib – avoid cutting into the intestines. Extend the incision so that you can view the abdominal organs in place. Note any abnormal contents in the peritoneal cavity, and take bacterial swabs if appropriate. Determine whether the organs are in their appropriate positions, but leave them in place at this point.
4. Cut through the diaphragm and remove the right half of the ribcage with the rib cutters or hacksaw. Examine the organs of the thorax, but leave them in place at this point, taking bacterial swabs if relevant.
5. Make cuts along the inside of the lower jaw, grasp and pull back the tongue. Cut the hyoid apparatus and draw back the tongue, oesophagus and trachea together to the level of the thoracic cavity. Remove the lungs and heart attached to the tongue, oesophagus and trachea, cutting attachments as you go. Sever the oesophagus and large blood vessels at the diaphragm. This group of organs is called the pluck – you must now examine it.
 - i) Examine the tongue and oral cavity.
 - ii) Dissect out the thyroid and parathyroids, and take tissue samples.
 - (ii) Palpate the oesophagus before opening it, looking especially under the mucosa in the part of the oesophagus that passes through the thorax for nodules caused by *Spirocerca lupi*, a nematode worm, which may sometimes grow large enough to obstruct the oesophagus.
 - iv) Examine the thymus, and take tissue samples.
 - v) There are also lymph nodes in the partition between the lungs, near the thymus: find one by palpation and take a tissue sample.
 - vi) Palpate the lungs, and note their colour and texture. Take a sample from the dorsal part of one of the apical lobes.
 - vii) Open the trachea and examine the contents. Extend the incision into the lung and through the bronchi.
 - viii) Open the pericardium (the fibrous sac that encloses the heart) and look for any abnormalities in the fluid. Take swabs if appropriate.
 - ix) Now examine the heart. There are several ways of doing this. The most important points are to examine all of the surfaces for haemorrhages, and all cut surfaces for pale patches. Look for lesions on the valves, and determine whether the size and shape of the heart is normal. Take samples from the septum between the ventricles, and from the papillary muscle (of the left ventricle).
6. Next, examine the organs of the abdomen. It is extremely important that you leave examining the intestines until last, because their contents are topologically outside the body and will, therefore, contaminate other tissues with bacteria from the outside world.
 - i) Remove and examine the spleen. Make multiple cuts through the parenchyma and take tissue samples.
 - ii) Remove and examine the liver. Make multiple cuts through the parenchyma and take tissue samples. Open up the gall bladder last, as the bile that it contains will damage the tissues. If the gall bladder appears thickened, sample it.
 - iii) Locate both kidneys and adrenal glands and remove them together. Cut the kidneys sagittally, peel off the capsule and examine all of the surfaces. Take tissue samples, ensuring that your samples include both the cortex and the medulla. Cut the adrenal glands in half, examine the cortex and the medulla, and take samples.
 - iv) Examine the bladder in situ before you open it. Have a vial ready to catch any urine, but only keep the sample if it appears abnormal. Take a tissue sample from the bladder.
 - v) Remove the stomach and the intestines, and cut all the attachments to separate the loops from one another. Take tissue samples from the pancreas and mesenteric lymph nodes. Then open the stomach and continue down the length of the gut to the rectum, taking tissue samples of the gut as you go. Bear in mind that the mucous membranes of the intestines are very easily damaged, so be careful, and never scrape the surfaces.
 - vi) Examine the reproductive tracts and take samples as necessary. Older domestic dogs often have tumours in the testicles which can be seen with the naked eye if you make repeated cuts through them.

Box 12.1 ... continued. Post-mortem examination of canids (adapted from Woodroffe *et al.* 1997).

7. It is always a good idea to look at the articulating surfaces of some of the joints. Open up the coxofemoral (hip) joints and look for abnormalities. The knees and the joints of the ankles and toes are also easy to look at.
8. Take samples of bone marrow by cracking one of the femurs near one end, and extracting a bit of the gelatinous marrow along with spicules of bone.
9. Perhaps the most crucial organ to sample in any dead canid is the brain, because many of the most important diseases that affect canid populations attack the brain.
 - i) Cut the skin and the neck muscles over the joint between the back of the skull and the first vertebra (the atlas).
 - ii) Bend the head forward to give access to the occipital foramen (the hole in the back of the skull).
 - iii) Push a drinking straw into the foramen and towards one of the eyes. In this way the brain stem, the base of the cerebellum, the hippocampus and parts of the cortex are all sampled.
 - iv) Before drawing back the straw, pinch it between your fingers to ensure that the brain sample does not fall back out of the straw. Then carefully withdraw the straw.
 - v) If you are storing your brain samples in 10% formalin, squeeze the brain sample out of the straw and into the formalin solution. If you are using glycerin solution, plunge the straw into the solution and cut the straw into pieces as necessary, but *do not remove the sample from the straw*.
10. Do not forget to collect samples for genetic analysis from the remains of the carcass.

patterns of infection are important. Closer and more direct contact between reservoir hosts and target species may be required for directly transmitted pathogens such as CDV or rabies, whereas indirectly transmitted pathogens such as canine adenovirus (CAV, urine-oral) or canine parvovirus (CPV, faecal-oral) can be transmitted when ranges overlap. When a pathogen (e.g., leishmania) is vector borne, the distribution and range of the vectors as well as host must be considered. The threat from diseases that tend to occur in epidemics is particularly difficult to assess. Even if reservoir surveys reveal no or low levels of disease, a large devastating epidemic could still occur.

The ecology of the carnivore system must also be considered. At one extreme, even if a disease that causes significant mortality occurs at high incidence in the reservoir, if there is no possibility of direct or indirect contact between reservoir and target species, then the risk of disease transmission and spill-over is effectively non-existent. Assessing the probability of transmission is surprisingly difficult, but a first step is to assess the geographical distribution of the different species and their degree of overlap (taking into account pathogen transmission mode). Inter-specific contact rates between members of the Carnivora have rarely been estimated (Rhodes *et al.* 1998). One exception is a study by Courtenay *et al.* (2001), in which they estimated contact rates between sympatric populations of domestic dogs and crab-eating foxes (*Cerdocyon thous*) in Brazil using radio-telemetry. The contact rate of individual foxes with the peri-domestic environment was calculated, as was the contact time in minutes. Potential exposure of each fox to the dogs in a village was calculated from this village contact rate and the mean density of dogs in the village. This approach found that foxes had a high level of contact with the peri-domestic habitats and that the probability of potential spill-over infections from dogs to foxes is high (Courtenay *et al.* 2001)

It is important to note, however, the uncertainty of extrapolating contact rates from healthy to diseased animals, particularly of directly transmitted pathogens. Changes in the behaviour of sick animals could in some cases either increase or decrease contact rates and transmission probability (e.g., rabies, Butler 1998). Further research is clearly required to explore this phenomenon.

12.3.5 Putting the seriousness of the threat into context

Wide-scale disease-related mortality in a population may clearly make a population vulnerable to extinction either directly or as part of the extinction vortex, through causing a population decline (Caughley and Gunn 1996). Disease may also be a limiting factor on population densities (for example, sarcoptic mange in red foxes in Scandinavia, and rabies in foxes in France/Germany), and thus depress population size (Lindström *et al.* 1994). A disease effect on fertility or recruitment rates could have a similar effect as one causing direct mortality (Caughley *et al.* 1992). Disease may thus cause a more insidious decline or prevent a population recovery after a perturbation, both of which are relatively hard to detect.

Although disease-related mortality is clearly catastrophic for the individual concerned, at the population level this mortality may be unimportant and have relatively little effect on the probability of population extinction. Thus, sporadic deaths may have little significance at the population level. That said, even sporadic deaths may affect population viability in very small populations, where pack sizes are small, where key members are affected or where the Allee effect might be important (Courchamp *et al.* 2000; Haydon *et al.* 2002).

By extrapolation, one-off epidemics, whilst clearly undesirable, may have little effect on long-term extinction probabilities, although they may have consequences for

loss of genetic diversity. The high reproductive capacity and short generation time of many canid species means that populations may recover relatively quickly (Lindström *et al.* 1994). In Botswana, when around 3% of ~70 packs of African wild dogs (*Lycaon pictus*) died out from rabies, the overall population size quickly recovered (J.W. McNutt pers. comm.). Similarly (although not canids), lions in the Serengeti recovered in four years from a 30% population decline due to CDV (C. Packer pers. comm.). However, recovery times may be hampered when the formation of new packs is a difficult and rare event.

When considering and prioritising management action to reduce the impact of disease, we need to try to quantify the risk that disease poses through mortality or through its effect on fertility or recruitment. A number of population viability assessments have attempted to incorporate disease mortality into their analyses (Ginsberg and Woodroffe 1997a; Vucetich and Creel 1999; Roemer *et al.* 2000b; Haydon *et al.* 2002). In general, these models have concluded that pathogenicity and canid population size are critical factors in determining whether disease has an effect at the population level. This inherent population resilience of larger populations reinforces the desirability of management strategies to maintain canids at the highest population size possible. Highly pathogenic infections such as rabies threaten the persistence of small- and medium-sized populations, with less pathogenic infections affecting only smaller populations. Other risk factors may include fragmentation (and thus where there is little possibility of recolonisation by dispersal from other areas) and populations with a high edge-to-area ratio, which will increase the probability of transmission from the reservoir species. In addition, species with low reproductive rates, low recruitment rates, where breeding units must be of a critical size to be viable, or where new breeding units rarely form, may also be more vulnerable to the effects of disease (Woodroffe and Ginsberg 1998; Courchamp *et al.* 2000; Haydon *et al.* 2002; Hudson *et al.* 2002).

12.4 Management of disease threat for threatened populations

The decision on whether to intervene when disease threatens canid populations will depend on a wide range of factors such as the prevailing conservation philosophy, logistical and financial issues, local cultural perceptions, and, of course, the degree of threat posed by a pathogen. Ultimately, this decision rests with local managers. If the decision to intervene is taken, a range of management options are available that have been outlined in various forms in a range of publications (summarised in Table 12.3, and adapted from Laurenson *et al.* 1997b and Woodroffe *et al.* 2004; see also Wobeser 1994). There is no universal panacea for managing disease threats to canids, and wildlife

managers should consider all options in their specific situation and weigh up cultural, logistical and financial considerations against the potential effectiveness of each approach. Here we discuss some of the issues surrounding the most important approaches in Table 12.3.

12.4.1 Direct protection and reduced transmission in target species

Direct intervention to cure or protect the target individuals has taken place in a number of situations. This approach may be one of the few options available to managers when the reservoir species is unknown or where a relatively intractable wild reservoir population is involved. As well as providing direct protection or treatment to individuals, this approach may also reduce transmission within the host population.

Treatment of individual Arctic foxes (*Alopex lagopus*) and foxes against sarcoptic mange has been carried out with apparent success (Morner 1992; Goltsman *et al.* 1996). In addition to the myriad logistical problems involved in administering treatments, particularly when a course of treatment is required, there may be relatively few situations where direct treatment can be used as few therapies are available for viral diseases, or that have been tested in wildlife.

Direct protection of target hosts through vaccination has also been used as a conservation tool in a number of outbreak situations, including African wild dogs, island foxes (*Urocyon littoralis*) and during the recent 2003/4 outbreak of rabies in Ethiopian wolves (*Canis simensis*) (Hall and Harwood 1990; Woodroffe 1999; Randall *et al.* submitted). Where vaccines are safe, effective and require little intervention to administer, when used prophylactically rather than in an outbreak situation, they have the potential to improve the viability of canid populations severely threatened by infectious disease. Population viability modelling suggests that vaccinating 20–40% of an Ethiopian wolf population against rabies could markedly reduce extinction risks, although in very small populations (25 animals) higher coverage is required to completely remove extinction risk (Haydon *et al.* 2002). This theoretical level of coverage is quite low in comparison with that required to eliminate rabies in reservoir hosts (70%, Coleman and Dye 1995). In this situation, rabies may cause an epidemic in a wolf population but vaccination protects a core of wolves that provide the basis for breeding and population recovery.

Attempts to protect wild dogs from rabies and canine distemper, acute threats to the persistence of small African wild dog populations, by direct parenteral vaccination have met with mixed success. The issues and controversy surrounding these attempts are extensively reviewed in Woodroffe and Ginsberg (1998) and Woodroffe *et al.* (2004), but the efficacy of rabies vaccines, particularly

Table 12.3. Management options for disease control for wild canids (taken from Woodroffe *et al.* 2003, which is modified from Laurenson *et al.* 1997b).

Option	Options	Advantages	Disadvantages	Likely benefits/ chance of success
Do nothing		Cheap, easy, evades controversy	Population viability not guaranteed	Depends on degree of threat
Reduce disease in reservoir species		No intervention with target	Must know reservoir. No guarantee of protection in target	
	1. Vaccination	1. Effective vaccines often available, particularly for domestic reservoir	1. May be extensive and expensive, logistical challenge if wild reservoir	1. May be high if wide <i>cordon sanitaire</i> and properly managed
	2. Culling		2. Cost, welfare, logistics	2. Low
	3. Limit reproduction/ ownership	3. May get to root of limitation of domestic reservoir population	3. Long lasting effect and species specificity not yet available. Difficult to change cultural attitudes to dog ownership	3. Reasonable adjunct in theory, but difficult to change attitudes. Other methods not currently feasible
	4. Treatment	4. Therapy availability depends on pathogen	4. Effective methods not yet available over large areas for most situations	4. Will depend on therapy availability and logistic resources, but low overall
Reduce disease in target species		Direct protection of target individuals	Handling sometimes required	Last chance in emergency situation
	1. Vaccination		1. Proven effective and safe vaccines for wildlife not always available	1. Can be high in some situations
	2. Treatment		2. Often unavailable or unfeasible	2. Depends on pathogen, better for sarcoptic mange
Prevent transmission between target and reservoir		No direct intervention	Must know reservoir	Higher on islands
	1. Fencing/ physical barrier		1. Often unfeasible	1. May be good if fences carnivore-proof
	2. Restraining domestic animal reservoir		2. Cultural constraints/conflict with dog function	2. Moderate to low
	3. Buffer zone (e.g., limit humans activities in protected areas)		3. Feasibility, cultural acceptability	3. Variable depending on feasibility

after a single dose, is questionable in this species and now the subject of further research. As with Ethiopian wolves, the feasibility of using oral vaccines warrants further investigation. Live oral rabies vaccines have been extensively used to control rabies in red foxes (*Vulpes vulpes*) in Europe and have been tested for potential use on both African wild dogs and jackals in southern Africa, although no field trial has yet been carried out (Bingham *et al.* 1999, Knobel *et al.* unpubl.). Preliminary trials suggest that an effective baiting system can be designed (Knobel *et al.* 2002). However, live vaccines present some safety concerns in both target and non-target species and it is difficult to carry out safety trials on all species that might consume the baits. For example, one strain (SAD, Street-Alabama-Dufferin) was protective for jackals, but induced clinical rabies in baboons (*Papio spp.*, Bingham *et al.* 1995), although other strains such as SAG2 are much more stable and have never caused disease in any species. The use of recombinant rabies vaccines, which incorporate only part of the rabies virus genome and cannot cause

rabies in target or non-target species, present a safer alternative from a rabies perspective for both target and non-target species (Kieny *et al.* 1984; Blancou *et al.* 1986). However, concerns over using genetically modified organisms or inadvertent contact with immunosuppressed humans and the vaccinia carrier may mean that some countries approach this method with caution.

Direct vaccination has also been used to protect island foxes from canine distemper in California's Channel Islands. Very high mortality associated with a distemper outbreak on Santa Catalina Island in 1999 demonstrated the need for a tool to protect recovering populations from distemper on this and the three other islands that had experienced major declines for other reasons (Timm *et al.* 2000). A new recombinant distemper vaccine, vectored by the canary pox virus (Timm *et al.* 2000) was tested, as this vaccine cannot replicate or shed CDV or pox virus in mammals. Trials on six captive foxes showed that the vaccine caused seroconversion with no observed side effects. Vaccination protocols were then trialed on the

western part of Santa Catalina Island not reached by the epidemic, which by then had faded out (S. Timm pers. comm.). Although, in the absence of challenge experiments, it is impossible to be 100% certain that vaccination confers protection from canine distemper, the existence of a distemper vaccination protocol known to be safe and likely to be effective for use in free-ranging island foxes is a valuable addition to the toolkit for the conservation of this Critically Endangered species (Woodroffe *et al.* 2004).

More recently, an emergency trial parenteral vaccination campaign was carried out to control an outbreak of rabies in Ethiopian wolves in the Bale Mountains (Randall *et al.* submitted). As permission had not been granted to test the efficacy of oral vaccines, wolves were trapped and vaccinated by injection with an inactivated rabies vaccine. Although this represented a huge logistical effort, preliminary results suggest good seroconversion rates but the trial is still ongoing. Only extensive monitoring work will enable the success or failure of this approach to be assessed.

This example notwithstanding, demonstration of the effectiveness of direct vaccination of target animals this approach has been limited (but see Hofmeyr *et al.* 2000, where vaccinated African wild dogs survived a rabies outbreak that killed other members of the pack). In most cases where direct vaccination has been employed, it has been a 'crisis' intervention dealing with acute disease risks where it has not been possible to leave unvaccinated controls. Overall, although this approach has some clear advantages (Table 12.3), vaccine and treatment availability is currently a severe constraint as few vaccines or treatments have been tested for safety and efficacy in wildlife. In addition, in the absence of challenge experiments in captivity, only situations where target hosts are naturally challenged will ultimately enable the efficacy of vaccines to be assessed. Feasibility studies are also needed to assess the population coverage levels that can be achieved using oral vaccines (Knobel *et al.* 2002). Nevertheless, during most disease outbreak this approach was essentially the conservation manager's only potentially feasible intervention option. Irrespective of their success, these interventions have yielded invaluable information on the logistics involved and efficacy of this approach.

12.4.2 Management of infection in reservoir hosts

Threatened canid species may also be protected from infectious disease through a reduction in the number of susceptible animals in the reservoir population, which will lead to a reduction or elimination of the disease and thus also a reduction in the chance of disease being transmitted to target hosts. A reduction in the number of susceptible animals can be achieved in several ways. First the overall number of animals in the reservoir population can be

reduced through limiting the population by culling or fertility control or, where domestic dogs are involved, by reducing dog ownership levels. Second, animals can be removed from the susceptible population by vaccination them. Clearly, we need to know which species is the reservoir of infection for this approach to be used. For many diseases affecting wild canids, domestic dogs are the reservoir host but wild canids have also been implicated in a number of cases. For example red foxes (Europe), yellow mongooses (*Cynictis penicillata*) (South Africa), raccoons (*Procyon* spp.), and skunks (*Mephitis* spp.) (North America) are examples of wild reservoirs for rabies, whereas a suite of wild carnivores may be involved in sustaining endemic canine distemper infection in Europe and North America. Control of diseases of public health concern, such as rabies or visceral leishmaniasis, has traditionally been carried out with this approach. The successes and failures of past culling and vaccination efforts thus provide important lessons for the conservation of rare canids threatened by infectious disease (Aubert 1994; Brochier *et al.* 1996).

Limiting host density by culling or fertility control

Where domestic dog populations are involved as a disease reservoir, culling or fertility control, although superficially attractive solutions, will not address the underlying demand for domestic dogs as these populations are usually limited by human decisions (Perry 1993). However, it ought to be possible, at least theoretically if not practically, to limit domestic dog densities to levels at which pathogens such as rabies cannot persist (e.g., Cleaveland and Dye 1995). However, cultural attitudes towards dogs vary widely, but in most developing countries their usefulness is acknowledged, for example as guards and cleaners. Limiting the number of dogs owned per family (either by law or changing social attitudes) may thus be culturally challenging to implement. Indeed, we know of no such successful programme. Moreover, where human densities are high, even comparatively low dog:human ratios may generate populations large enough to represent a disease risk to local wildlife and thus the success of this approach can be limited.

Culling wild canids such as foxes to control rabies, although sometimes successful in the short term in a limited area, has otherwise met with failure due to the rapid recovery of fox populations and the continued (expensive) culling effort required (Macdonald 1993; Aubert 1994). In addition, changing moral attitudes are rendering this approach obsolete. Culling domestic dogs, although frequently used to remove problem or un-owned urban dogs in many countries, is also becoming less acceptable, at least in the north.

Fertility control could also be used to manage canid population densities and, in theory at least, show some promise (Barlow 1996). In practice, given that female dog

sterilisation by surgery is expensive and culturally and logistically difficult, and that dog populations are rarely closed, this approach may be limited in its success. Fertility control would be even more difficult to achieve among wild canids, although initial investigations of immuno-contraceptive vaccines, which target the release of reproductive hormones, have shown encouraging results for red foxes in France and Australia, and are being developed for domestic dogs (Fayer-Hosken *et al.* 2000). Oral chemical contraceptives are available for use in wildlife (reviewed in Tuytens and Macdonald 1998) but, like poisons, their use in areas occupied by threatened populations would be inappropriate. Despite these concerns, immuno-contraception – especially if it could be combined with vaccination – may be a feasible option if technology can be developed for the future management of disease reservoirs.

Vaccination of reservoir hosts

Experience from the control of rabies risks to humans and livestock suggest that vaccination of both domestic dogs and wild canids may be powerful tools for the protection of threatened species from acute disease threats. Direct vaccination of wild reservoir hosts (foxes, raccoons, coyotes, skunks) has successfully reduced the incidence of rabies on a large scale in Europe and in the United States (Brochier *et al.* 1996). Moreover, large-scale vaccination of domestic reservoirs is commonly conducted for a variety of reasons in the animal health sector. In the last decade this approach has been adopted in several situations in an effort to protect wild canids. Programmes have attempted to create a *cordon sanitaire* in the reservoir around threatened populations. In theory the area to be covered will vary considerably with the density of the canid of interest: the same size population of African wild dogs (home range 400–1,200km² per pack, Woodroffe *et al.* 1997) than Ethiopian wolves (home range 6–11km² per pack, Sillero-Zubiri and Gottelli 1995a). As threatened canids are generally surrounded by a sea of reservoir hosts, whether wild or domestic, regional eradication is near-impossible without a wide-scale coordinated rabies control programme. As such, without national or international control programmes, it is likely that vaccination cover would have to be maintained in perpetuity to control the disease threat. However, a coordinated rabies control programme with both public health and livestock authorities would reduce the cost borne by the conservation community and both financial (primarily from a reduction in livestock losses) and public health benefits would accrue to local populations. Vaccination of domestic dogs by wildlife managers also provides benefits that may improve protected area-community relations as both parties may cooperate and spend time in a mutually beneficial activity (Sillero-Zubiri and Laurenson 2001). As such it can be a powerful tool for

wildlife managers looking for opportunities for cooperation and communication.

In rural Tanzania, vaccination of domestic dogs has shown that a simple central-point vaccination strategy, resulting in vaccination of 60–65% of dogs against rabies and distemper adjacent to Serengeti National Park, has significantly reduced the incidence of rabies in dogs and risk of exposure to people, with opportunities for transmission to wildlife also decreasing (Cleaveland *et al.* 2003). Dog vaccination campaigns have also been carried out around other national parks in Tanzania such as Ruaha, Arusha and Tarangire. In the Bale Mountains National Park in Ethiopia, intensive dog vaccination against rabies and CDV started in 1998 within the park and where resources allowed, in neighbouring communities. No case of these diseases was reported between 1998 and August 2003. Rabies cases in dogs and other species still occurred at the edge of vaccination zones, although the overall incidence in dogs and humans very much reduced (Ethiopian Wolf Conservation Programme unpubl.). However in September 2003, rabies broke out in Ethiopian wolves in one area of the park, thought to have been brought in by an unvaccinated immigrant domestic dog (Randall *et al.* submitted). It is thus apparent that a wide *cordon sanitaire* is required, particularly when transhumance of humans and their domestic animals occurs. This outbreak also illustrates a disadvantage of this approach: there is no direct protection of the target species and success cannot be guaranteed if intervention is carried out on too small a scale and where there are inadequate resources to cover such areas. Furthermore where payment for vaccination is expected or where dogs are used for illegal hunting are not presented for vaccination, the success of the approach may be curtailed.

Concern has been expressed that vaccination of disease reservoirs – especially domestic dogs – could remove an agent of population limitation and lead to increased host density (Moutou 1997). This could be potentially damaging, especially if vaccine cover were to be halted (Woodroffe 2001). However, dog populations are thought to be limited by humans (Perry 1993) and preliminary studies indicate that, while dog vaccination in northern Tanzania has led to a significant decline in mortality rates, population growth rates have not increased (S. Cleaveland unpubl.). It appears that the reduced demand for pups has lowered recruitment rates, resulting in a much more stable population which is protected against rabies. However, data are still required to assess longer-term demographic impacts and the demographic impact of mass vaccination in different settings.

Overall, although this approach is not without its problems and requires considerable financial and logistical resources (Table 12.3), it is currently the most feasible method available to managers where a domestic reservoir is involved.

12.4.3 Reducing disease transmission between host species

In theory, limiting disease transmission between reservoir and threatened hosts should be effective in reducing the threat of disease. Reduced transmission could be achieved by eliminating range overlap between the species; that is by clear physical separation. Indeed bighorn sheep (*Ovis canadensis*) have been protected from pneumonia and scabies transmitted from domestic sheep (*O. aries*) by barring domestic sheep from buffer zones surrounding bighorn populations (Jessup *et al.* 1991).

In gazetted national parks there are few reasons to tolerate domestic dogs and unaccompanied dogs are usually captured or extirpated. In areas where separation is not possible, the control of free-ranging domestic dogs presents a substantial challenge and may be near-impossible in many situations. For example, game fences around wildlife areas in South Africa should reduce contact between wild canids inside and wild or domestic canids outside. Indeed, fences around Kruger National Park may partially explain the absence of evidence of exposure to CDV and canine parvovirus among wild dogs (Van Heerden *et al.* 1995). However, small carnivores are notoriously adept at crossing such fences and a game fence did not prevent rabies killing wild dogs in Madikwe Game Reserve, probably transmitted across the fence by jackals (Hofmeyer *et al.* 2000).

Where ranges of target and reservoir hosts do overlap, there may still be measures that can be taken to reduce disease transmission. Limitation of contact between Ethiopian wolves and sympatric dogs inside wolf habitat has been attempted through community education programmes that encourage people to tie up their dogs at home. Collars and chains were supplied, but older dogs were adept escape artists and collars and chains were very useful for other purposes, such as tying up calves or horses (Sillero-Zubiri and Laurenson 2001) is probably almost impossible to accustom older dogs to accept being tied up, thus training would have to start with the next generation of pups. In addition, cultural resistance from dog owners is likely to hinder such efforts, not least because restricting dogs may reduce their usefulness as guards and cleaners, and their ability to forage when people cannot afford to provision them. In addition, cultural taboos against close contact with dogs are present in some areas, again increasing cultural resistance to handling and tying. Overall, the success of this approach may be limited and it must be recognised that cultural change occurs slowly, both in terms of the generational times of humans and dogs.

12.5 Which approach is best?

Infectious disease is a threat to wild canids that conservationists are ill-equipped to manage. Lack of information hinders management of this newly recognised threat – there are no established models to follow, and some early and unsurprising failures have attracted damaging controversy (Woodroffe 2001). This makes it difficult to assess which approach is most likely to meet with success. However, it is important to recognise that the decision not to intervene is in itself a conscious decision and not a lack of consideration. In general, intervention will be most warranted in small and isolated populations with a high edge-to-area ratio and for highly pathogenic infections (Woodroffe *et al.* 2004). Intervention may be less warranted in larger populations and for less virulent infections. However, the decision to intervene may be taken due to local political and cultural considerations or where additional public health or economic benefits accrue to local communities. Where intervention is warranted, vaccination of threatened hosts or of domestic reservoir hosts presents the most feasible options in our disease management toolkit at present, particularly when safe, effective and practicable vaccination protocols are available. Vaccination of wildlife reservoirs is more problematic. In all situations, the specific conditions in the area will determine what action or actions are taken as there is no universal panacea to mitigate the threat of disease for canids.

12.6 Conclusions

Disease threat assessment and management for wild canids often presents a challenging issue for managers with the apparently complex ecology and epidemiology sometimes involved. Field managers must take a multi-disciplinary approach and bring in the expertise of veterinarians, ecologists, theoreticians and others. The Canid Specialist Group's Disease & Epidemiology Working Group is also always available and willing to give advice. Improved surveillance is fundamental to improving management and further testing of alternative management options is required. Informed decisions cannot be made without such background information. Although management options may often ultimately depend on non-biological factors such as feasibility, the cultural or political environment or the availability of finance, we should strive to take objective and scientific management decisions with the best information available.

Management of Wild Canids in Human-Dominated Landscapes

C. Sillero-Zubiri and D. Switzer

13.1 Introduction

Whereas some canid species are declining globally under the pressure of habitat degradation and fragmentation, disease and persecution, other have managed to survive and even thrive in human-dominated landscapes. This overlap with people results in competition for resources, which is at the heart of the conflict between many wild canids and man.

Canids prey upon a range of livestock, game stock, and threatened wildlife, and a few of the larger species may also attack, and on rare occasions, fatally injure humans. As a result of this and innate human prejudice, canids have frequently been vilified by people, classed as vermin and actively sought out and killed. Control may be occasional and opportunistic, such as when a farmer shoots a fox attacking his chickens, or targeted and systematic, such as with control programmes to reduce or eradicate problematic species. Harassment notwithstanding, canids have survived in many areas where other carnivores have gone extinct. Their resilience is principally due to their relatively high reproductive rate (i.e., large litter sizes and early sexual maturity), which compensates for increased human-inflicted mortality rates, and their adaptability to new environments. Canids can often quickly recover from population decreases and range contraction, and rates of re-colonisation are often high due to high levels of dispersal (Gittleman 1989).

This chapter examines the reasons why canids frequently find themselves in conflict with humans, and the ways in which conservation biologists and managers have tried to solve these problems. Data for this chapter are derived from original research, literature reviews, and from a preliminary analysis of the information provided in the species accounts of Part I of this action plan. This paper borrows several ideas first developed by Sillero-Zubiri and Laurenson (2001) for carnivores in general. For additional information on managing conflict in canids, see Sillero-Zubiri *et al.* (2004b).

13.2 Why do canids come into conflict with humans?

Given the long tradition of canid persecution, it is pertinent to ask what the reasons are behind human-canid conflict.

Although in recent times public perception of canids has improved, historically, the majority of people have held negative views towards wild canids, with these views being handed down through the generations, and then carried with them as they migrate. These views have often been generated by an ingrained fear of the larger species, and by the recurrent issue of wild canids preying on livestock and valuable game species. Europeans, for instance, took their intolerance of carnivores with them when they colonised other regions of the world, notably Africa and America. In North America, grey wolves (*Canis lupus*) and coyotes (*Canis latrans*) were actively hunted by the colonists upon their arrival, and by the 1930s wolves had been extirpated from most of the USA, with only Minnesota and Alaska harbouring viable populations (Mech 1970). Elsewhere, Europeans were responsible in 1876 for the last known canid extinction, namely that of the Falklands wolf or Malvinas fox (*Dusicyon australis*) (Macdonald and Sillero-Zubiri 2004b). Similarly, African wild dogs (*Lycan pictus*) were considered vermin by European farmers and deliberately killed on farmland and protected areas (Woodroffe *et al.* 1997).

Intriguingly, Africans' perceptions may differ from those of Europeans; for example, wild dog kills are considered a useful source of meat by the Shona in Zimbabwe (G. Rasmussen pers. comm.), and the Maasai people of East Africa regard them as an asset as they prey on wildebeest (*Connochaetes taurinus*), which they regard as competition for grazing with their cattle (Fuller and Kat 1990).

13.2.1 Attacks on humans

The larger canids are often considered a direct threat to human life. Grey wolves and wild dogs, and to a lesser extent dholes (*Cuon alpinus*) are portrayed as villains in the myths and folklore of many cultures (Landau 1993). Attacks on humans are quite rare though, and deaths are even less frequent (Linnell *et al.* 2000), yet many people still hold a deep-rooted fear of wild canids and this contributes to the overall negative view held in many societies. For instance, attacks by wolves are still feared by people in the USA, even though, in contrast to other predators like bears and mountain lions (*Felis concolor*), there have been no verified attacks in the last century

(Mech 1970; Kellert *et al.* 1996). African wild dogs are often described as ‘ruthless killers’ (e.g., Bere 1955), although attacks on humans rarely, if ever, occur (Creel and Creel 2002). In situations where canids do attack people it is often due to an individual rabid animal in ‘furious’ phase, attacking people repeatedly over the short time period they survive (Linnell *et al.* 2000). With the eradication or reduction of rabies in many parts of the wolf’s range, the incidence of wolf attacks has dropped dramatically, but cases are still reported from Asia and the Middle East (see Chapter 12).

The majority of present-day predatory (non-rabies) wolf attacks seem to occur in highly modified environments, with little or no natural prey, with wolves habituated to people presence and relying heavily on human refuse and livestock as an alternative source of food. As many as 273 children have been reported killed by wolves in the last 20 years in northern India, where wolves come into close contact with shepherd children due to their dependence on villages for food (Jhala and Sharma 1997). Wolf attacks may result from habituation, with wolves losing their fear of humans and increasing the likelihood of encounters. Furthermore, attacks may also result from provocation when wolves are cornered, trapped or a den with pups is breached.

Although attacks are very rare, when they occur they attract a disproportionately large amount of sensationalist media coverage, and have the potential to be very damaging to canid conservation efforts. For example, the recent killing of a young boy by a dingo (*Canis lupus dingo*) on Australia’s Fraser Island has led to calls for a dingo cull, even though this population is considered one of the most genetically pure dingo populations remaining (Queensland Parks and Wildlife Service 2001). Thankfully, it appears that the efforts of conservation groups are proving increasingly successful at overturning this inherent human fear of canids, and conservation support for wild canids is gaining in popularity.

13.2.2 Competition for resources

Ever since our ancestors began herding livestock, many carnivore species have been persecuted for their role as predators of domestic animals. Throughout Europe there were, and in some instances still are, deliberate policies to reduce the numbers of predatory species to safeguard livestock and poultry, and also to promote human safety and to benefit game species. Certain canid species have experienced a long history of organised persecution; for example, in 6th century BC Greece, the city of Athens issued state bounties for the killing of wolves in an attempt to protect livestock from predation (Reynolds and Tapper 1996). Grey wolves were also deliberately exterminated from the islands of Japan, except Sakhalin, even though other large mammalian carnivores were allowed to remain

extant (Dobson 1994). In some areas predator reduction was so effective that canids survived only in very low numbers or were completely extirpated. For example, in Britain grey wolves, brown bears (*Ursus arctos*) and lynx (*Felis lynx*) were all extinct by the middle of the 19th century (Harris 1989).

Unlike the exaggerated problem of attacks on humans, canid predation on livestock and game is a harsh reality difficult to deal with. In many countries large-scale control policies have traditionally been deployed to reduce livestock predation, and are often written into state law, and rewarded by bounty schemes (Childes 1988). Yet, the resources and expense invested in control campaigns have often been disproportionate with the damage perpetrated. For example, in South Africa £400,000 (currency of the period) was spent between 1915 and 1925 to extirpate predators (Pringle 1977).

More recently, conservationists have led a re-examination of the costs to farmers, hunters and society as a whole, of preserving viable carnivore populations (e.g., Clark *et al.* 2001; Sillero-Zubiri and Laurenson 2001). Although the general public is no longer willing to see wild canid populations reduced simply because they come into conflict with certain sectors, this change is not necessarily taking place among people living near wild carnivores. As a result, canids still face substantial persecution, particularly in those areas where they spill-over from the edges of protected areas (Woodroffe and Ginsberg 1998). Meanwhile, conflict has not been a problem for many decades in agricultural landscapes where carnivores have been extirpated, but this situation is changing as many carnivore populations, especially wolves and coyotes, have been increasing and reclaiming some of their historical ranges (Mladenoff *et al.* 1995).

Predation on livestock

Every domestic species, from chickens to cattle, is affected by canid predation. Indeed, livestock predation was the most frequently cited reason for problems between humans and canids in part 1 of this volume. Due to selective breeding and protection, domestic animals exhibit little effective anti-predator behaviour, making them particularly vulnerable to predators (Kruuk 2002), with domestication effectively breeding out the “wildness” of these animals. Changes in livestock husbandry and a decrease in the manpower employed in livestock production, most rapid and advanced in the developed world, have resulted in animals rarely herded (e.g., Rasmussen 1999) or guarded by dogs and thus more vulnerable to predation. In addition, livestock compete with wild herbivores for resources and, subsequently, they can either reduce the abundance or alter the distribution or behaviour of wild prey, thus changing the pattern of predation to include more livestock (Yalden 1996). Although farmers consistently express the most negative

attitudes toward large canids, such as grey wolves and African wild dogs, they often constitute a minor problem compared with smaller canids such as jackals, coyotes, and feral dogs (e.g., Ciucci and Boitani 1998).

Livestock losses can be significant. Estimates of sheep losses to wild canids in the USA, for example, ranged from US\$19–150 million between 1977 and 1999, and cattle losses represented US\$52 million in 2000 (Knowlton *et al.* 1999). Although these losses represent only a fraction of the total income generated by the USA livestock industry, often they are not spread out evenly over the farmer community but rather are borne by a few individual farmers who endure the majority of the damage. In developing countries high livestock losses can have a serious impact on farmers' livelihoods. In India's Spiti Region losses to predators averaged US\$128 per family per year in three villages, which equates to half the average annual per capita income (Mishra 1997).

Predation on game species

Throughout history, canids have been seen as competitors for prey and this remains the second most significant cause of human-canid conflict today. In Europe, gamekeepers target red foxes and other predators in an attempt to increase the population of partridges, pheasants, and grouse that are available for hunters to shoot (e.g., Macdonald *et al.* 2000). In North America, carnivore populations have traditionally been controlled in order to increase game species, in particular wild ungulates. This strategy was prevalent even amongst conservationists, and predator removal was the National Parks' policy until the latter half of the 20th century (Clark *et al.* 1999). Although carnivores are increasingly valued as an integral part of the ecosystem, and in spite of the fact that canids mostly target the sick and infirm animals that are not sought after by hunters, there is still great pressure from the hunting lobby to reduce their numbers. In Alaska, for example, grey wolves are blamed by hunters for declining moose and caribou populations and the resulting reduction in hunting quotas (Gasaway *et al.* 1992).

Predation on threatened wildlife

Wild canids can also have a detrimental effect on other wildlife, particularly where they have been introduced to isolated islands, where ground-nesting seabirds have often not developed any defence mechanisms to avoid predation. Arctic foxes (*Alopex lagopus*), for instance, have been responsible for large reductions on several Arctic seabird colonies, due to either being introduced by man (Bailey 1993) or where they have naturally invaded islands (Birkhead and Nettleship 1995). See Courchamp *et al.* (2003) for a review of mammal invaders on islands.

In a few exceptional circumstances, there may be a novel conservation dilemma when a threatened canid species has had a significant impact on another threatened

species. In the Alas Purwo National Park, Java, Indonesia, dholes were deemed responsible for the decline in the Endangered banteng (*Bos javanicus*) (Indrawan *et al.* 1996), and consequently a reduction in their numbers was recommended. Asian wolves (*Canis lupes pallipes*) in Velavadar National Park, India, may play a part in limiting the population size of the Vulnerable blackbuck antelope (*Antelope cervicapra*) (Jhala 1994). While these conflicting situations might only pose a localised threat, they do create a conservation quandary when ecosystem conservation, practical management and animal welfare must all be balanced with existing land-use.

In a reversal of the above scenario, threatened canid populations can also be under threat from other species, such as African wild dogs facing competitive exclusion from spotted hyenas (*Crocuta crocuta*) and lions (*Panthera leo*) in some protected areas (Creel and Creel 1996). Canid populations can also be threatened through intra-guild competition, such as with coyotes preying heavily on sympatric swift foxes (*Vulpes velox*) (e.g., Carbyn *et al.* 1994).

13.3 Solving problems: approaches to canid-human conflict

The traditional way to deal with conflict between humans and wild canids was to attempt to remove the threat simply through extermination. This approach of blanket predator control has traditionally been the backbone of canid management, with farmers and wildlife managers trapping or poisoning canids wholesale. However, the outcome of indiscriminate control is not always straightforward, both in technical terms and due to widespread resistance amongst the general public.

Widespread killing seldom delivers effective long-term predation reduction and the preferred approach is now one that focuses on changing the behaviour of the individuals directly involved in damage, and also addresses the behaviour of the people that are facing the problem. Non-lethal methods are increasingly favoured to prevent, or at least reduce, the incidence of predation, and the management goal is slowly shifting to minimise impact on innocent individuals, while minimising human-canid conflict (Treves and Karanth 2003). Indeed, in some instances the predation problem can be exacerbated where culling allows the immigration of problem animals into vacant territories (Sacks *et al.* 1999). Consequently, removing culprit individuals from a canid population may be more efficient than attempting population control (Conner *et al.* 1998; Blejwas *et al.* 2002).

In the next section, various lethal and non-lethal methods are examined, followed by a review of approaches that attempt to reduce or eliminate human-canid conflict by increasing human tolerance towards wild canids.

13.3.1 Predation control and reduction

Trapping

Trapping is perhaps the oldest method used to reduce predator numbers and a wide variety of cage, box, leg-hold traps and snares have been used, either killing the malefactor directly or holding it until it can be destroyed by a returning hunter. Traps are often unselective and may kill non-target species (see Conover 2002). Traps have been made illegal in many parts of the world due to concerns over the high level of stress, pain and suffering on the animals that are caught. Notwithstanding this, trapping is still a major method for canid reduction, most notably in the control of coyotes in the USA and Patagonian foxes. Most countries where commercial trapping for furbearers still occurs, e.g., Canada, the European Union and the Russian Federation, have regulated open and closed seasons and restrictions on methods of capture, under an agreement on international trapping standards signed in 1997. Recently, an ISO standard was developed for mammal trap testing (ISO 1999), and there are efforts to research and disseminate best trapping practices (e.g., IAFWA 2003).

Shooting

Perhaps the most widely used method to kill canids, shooting is labour intensive but species-specific. For certain species, e.g., red fox, shooting with a rifle is usually regarded as offering the best combination of efficiency and humaneness, and is often carried out at night with a spotlight and vehicle (Reynolds and Tapper 1996). Shooting on a large scale has been used in North America to control canid populations, including aerial hunting from helicopters; this technique is commonly used by agriculture agencies in the western USA to reduce coyote predation on sheep (Wagner and Conover 1999). Used in combination with expert tracking, or stalking at a kill site, shooting becomes a good method for targeting problem animals, although it requires experienced personnel.

Denning

Farmers often resort to trapping canids in their dens, digging them out and euthanising them, or fumigating the den and asphyxiating the occupants. Although still legal as a means to control pest species in the USA and elsewhere, both methods are considered inhumane and their use is strongly discouraged.

Poison baiting

Baits containing poison are often used in schemes to eradicate canids from a large area, and were one of the chief methods by which wolves were exterminated from large parts of Europe and North America (e.g., wolf poison campaigns to increase wild ungulate populations in Alberta, Canada in the 1950s and 1960s; Gunson 1992).

Among population culling techniques, poisoning exemplifies a necessary trade-off between utility (cost-efficiency), conservation (target-specificity) and humaneness, with no one method satisfying all criteria (see Sillero-Zubiri *et al.* 2004b). In addition to widespread opposition on welfare grounds, there are serious concerns about the effects of toxins on other wildlife and livestock, since poison baits are not discriminatory. This is of particular concern to conservationists where the intended target species is sympatric with populations of a threatened species. In order to circumvent the specificity shortfall, coyotes in the USA have been targeted by a spring-powered device called M-44 that delivers a lethal dose, and that is selective by using bait designed only to attract canids (Beasom 1974).

Livestock protection collars

This collar consists of a sachet of poison attached to the neck of the domestic animal needing protection. It operates on the premise that many canids kill by a neck bite, and they would thus puncture the sachet, release and swallow the deadly poison. These collars are particularly effective as they target only the individual canids that are responsible for killing livestock, sparing those that do not engage on livestock predation, and take individuals that have evaded other capture methods (Burns *et al.* 1996). One study in California showed a halving of lamb losses to coyotes from 15.8 % to 7% of the flock (Timm 1999). The main advantages of this method are its high specificity, its potential application on other livestock including cattle and goats, and protection against a number of predator species. Unfortunately, the need to equip most of the herd or flock with collars renders it impractical and expensive. Additionally, there has been objection over the use of the compound 1080 (sodium fluoroacetate) in the collars and the effect it may have on non-target species, as collars may accidentally fall off and subsequently come in contact with other wildlife.

Sport hunting

Hunting canids for sport remains a traditional pastime in Europe, Asia, North America, Patagonia and Australia. It may include hunting with firearms, bows or crossbows, or large organised hunts using horses and packs of dogs. From a management perspective, sport hunting can be used to offset livestock losses or dispose of known 'problem' animals, while it may also be useful in conserving populations of canids since it can increase their value. For example, some foxhunts in Britain have actually invested in management policies that conserve a certain population level of red foxes, such as habitat creation, artificial den sites, and artificial feeding, in order that there is sufficient quarry to hunt (Reynolds and Tapper 1996). Notwithstanding this, it is unlikely that sport hunting would deliver a viable alternative for mitigating human-

canid conflict, as most canid species are not sought after by sports hunters.

13.3.2 Non-lethal alternatives

The search for non-lethal alternatives to manage canid conflict has intensified over the last few decades due to increasing conservation and welfare concerns. Non-lethal approaches should not be thought of as a completely novel area, as many such methods (for example, suitable husbandry, guarding dogs and barriers) were traditionally used to reduce canid predation on livestock and game. Unfortunately, the traditional methods declined in use with the intensification of agriculture, but there is now a move back towards some of these traditional techniques and also a search for new non-lethal techniques. The most important aspect to realise with the development of these alternatives is that there is no one method that would be applicable in all situations, and often several may be needed in combination to significantly reduce conflict.

Use of fencing as a barrier for predation

Fencing can be employed either to keep predators out of a particular pasture, field or enclosure containing valuable stock, or to keep them confined within a particular area, such as a wildlife reserve. In Africa there is widespread use of traditional *bomas* or *kraals*, small enclosures built from dense thickets of Acacia bush (*Acacia* spp.), to protect livestock from predators when they are most vulnerable (e.g., at night or during calving). Kruuk (2002) showed that a simple thorn-bush boma could make a large difference to subsistence shepherds in northern Kenya, where 90% of all losses to predators took place outside enclosures. In Europe and Asia, livestock has traditionally been fenced in by hedges, stone-walls, wooden fences and more recently, barbed wire. Although these barriers are effective in preventing animals from straying, they offer little protection from predation, as they are permeable to discerning carnivores.

Small predator-proof fences have been used to protect ground-nesting birds (Bailey 1993), and have successfully excluded Arctic foxes from the nests of Alaskan pectoral sandpipers (*Calidris melanotos*) (Estelle *et al.* 1996). Installation costs and maintenance of predator-proof fences tend to be prohibitive at a large scale, and fencing would be impractical to prevent canid predation on sheep production systems in the western USA and Argentine Patagonia (Knowlton *et al.* 1999). In Australia, however, a 5,614km fence excludes dingoes from sheep farming lands in South Australia, Queensland, and New South Wales (Reynolds and Tapper 1996), and it is deemed that sheep farming would not be viable without this fence.

Electric fences provide a promising non-lethal predation avoidance/protection system that also protects

the carnivores involved, and can be cost-effective for some species in some situations (Balharry and Macdonald 1999). In the Rumanian Carpathos, tests of mobile night corrals were successful at significantly reducing losses to wolves and bears (Mertens *et al.* 2002). As a cheaper alternative to wire fences, Musiani and Visalberghi (2001) propose “fladry”, a line of red flags hanging from ropes traditionally used to hunt wolves in eastern Europe, which in tests showed captive wolves avoiding the flags even when the daily food ration was placed on the other side.

Fencing reserves and their wildlife has been used as a way of reducing conflict with the surrounding communities. Although this is an outdated conservation approach and antagonised with modern ‘open’ conservation systems, in many places it has proved very effective. Several public and private areas in South Africa are prime examples, such as Kruger or Pilanesberg National Parks. Unfortunately, permanent fence construction and upkeep are costly, thus precluding their use in poorer countries. More importantly fencing effectively cuts wildlife movement and may result in catastrophes during droughts or bushfires, and the small size of many fenced reserves means that populations of canids with a small genetic pool will require active management.

Improving livestock husbandry

Predation risk tends to increase with herd size, distance from people and buildings, proximity to thick cover, and carcasses left in the open (e.g., Mech *et al.* 2000; Kruuk 2002); many of these attributes are brought about by the intensification of livestock farming.

Diligent husbandry is essential to prevent unnecessary losses, such as improved vigilance, preventing livestock from straying and returning herds to enclosures at night (e.g., Kruuk 2002). It has also been argued that in certain economies, utilising additional manpower is justified because it has economic benefits beyond those accrued simply through reduced predation. Most of these are resultant from improved stock tractability and herder vigilance, and include reduced stock theft, increased weaning weights because calves spend more time with their dams, and sick animals or cows with calving difficulties are noticed earlier. In addition, as cattle become more manageable with the continual handling there is a reduction in losses to physical injuries from breakouts and they benefit from fewer stress-related problems (Rasmussen 1999).

Specific husbandry practices, however, must be developed for the particular situation of each producer’s group, and evaluated accordingly, to prevent the use of practices that may only delay predation or have undesirable side effects (Knowlton *et al.* 1999). At a larger land-use scale, diversification has been proposed to reduce conflict (Johnson *et al.* 2001), such as shifting from sheep to cattle husbandry (e.g., Patagonia).

Livestock guarding animals

Livestock guarding dogs (LGDs) have been used by shepherds to guard their flocks from predator attacks in Europe and Asia as early as the 6th century (Rigg 2001). With the foregoing of traditional livestock techniques, the use of LGDs had been in decline in much of Europe, and flocks were left unprotected in many areas. However, the recovery of predator populations in many areas has led to a re-awakening of interest in using LGDs to protect livestock. For instance, the reintroduction of LGDs is currently underway in several southern and Eastern European countries to prevent wolf predation. LGDs have also been extensively trialed in the USA and are now in use in a number of western states to help reduce coyote predation on sheep.

A good LGD is usually large, independent, intelligent, attentive and gentle towards livestock, but aggressive towards predators (Knowlton *et al.* 1999). The dogs are placed with a flock or herd of animals from an early age and bond with them, effectively becoming part of the herd. They remain with the herd at all times, even when humans are absent, alerting the flock and shepherds to the presence of predators and will themselves attempt to drive predators away from the flocks. Overall, the economics of using LGDs is dependent on a number of factors including the annual rate of predation, the ability and longevity of the dog, and the costs of purchase and maintenance (Green *et al.* 1984). To be effective LGDs must be able to see predators approaching easily, and, therefore, it is best to use them in flocks of 100–200 sheep in large open pastures, or instead in small fenced areas (Rigg 2001).

Although the most frequently used, dogs are not the only animals that can be used for livestock guarding. Llamas have proved to be effective livestock guards in certain situations and since the early 1980s have been increasingly used in the USA to defend against predation on sheep by wolves and coyotes (Meadows and Knowlton 2000). Llamas have an inherent dislike of canids, and when pastured away from other llamas will bond with sheep becoming part of the flock. They have several advantages over LGDs, namely they live longer, require less training, have a faster acquisition of guardian status, have fewer special management considerations such as food and maintenance, and are more compatible with other depredation techniques (Meadows and Knowlton 2000). However, their ability to guard the sheep depends on the ability to see the whole pasture and the sheep within it, and, therefore, it may be better for them to be used in relatively small, flat, open pastures (Timm 1999). They can also cause trouble if dogs are used to herd the sheep, as they will often behave aggressively towards these dogs (Timm 1999).

Donkeys were often used to defend livestock from predators in Namibia when European-owned farms developed there a century ago, and are now making a comeback (Rigg 2001). In Switzerland, donkeys have been

used to guard sheep since 1995. Donkeys are capable of providing a high level of protection at a relatively low cost and level of maintenance (Rigg 2001), and their use could be very beneficial in developing countries where the cost of maintaining LGDs may be too high to be economically viable for most farmers.

Translocation of problem animals

Translocation has been used in North America to manage individual grey wolves involved in livestock depredation. It has also had limited application with African wild dogs, involving the relocation of whole packs from problem farmland areas. In Zimbabwe such translocations have showed high survival of the new founder stock (90% survival), followed by successful reproduction in the new area. This experience has showed promise for similar translocations in the future, as it delivers a face saving excuse to farmers that otherwise would have illegally killed the dogs. Even though new dogs have slowly filled the artificial vacuum created by the translocation, the latter has assisted to maintain an '*entente cordiale*' with landowners (G. Rasmussen pers. comm.).

There are concerns that the survival of translocated animals may be poor, particularly in an ecosystem with a high density of conspecifics where a translocated animal could get an opportunity to fit into the social system (Linnell *et al.* 1997). For example, of 107 wolves translocated in northern Minnesota following depredation or harassment of livestock, 17% were shot, or recaptured at least once, for re-offending (Fritts *et al.* 1984). Although the mortality of translocated wolves was not higher than that of resident wolves, pack mates failed to stay together and travelled long distances with some animals returning home. It would appear that unless there are large areas available with a low density of conspecifics and where conflict potential is low, this strategy is unlikely to work (Linnell *et al.* 1997) (see Chapter 15 for a more detailed discussion of canid translocations).

Conditioned taste aversion

The principle of conditioned taste aversion (CTA) relies on taste and olfactory agents that cause a deep and lasting aversion to associated tastes and create a negative association between eating a particular food and sickness. Gustavson *et al.* (1976) first suggested that lacing lithium chloride on baits could be a useful management tool for problem predators and extensive trials have been conducted since, but because of poor experimental design, results have been equivocal and, therefore, controversial (Reynolds 1999). Unfortunately, CTA does not seem to be viable as a canid deterrent (Andelt *et al.* 1999; Linnell 2001); predators do not seem to associate the illness with killing behaviour, and, therefore, they often continue to kill livestock. Furthermore, there are practical problems with CTA application, in that it requires the predator to be treated

several times, and as new individuals are recruited into the population treatment would need to be repeated regularly.

Direct, generalised aversion to foul-tasting substances, such as Bitrex™, has an advantage over CTA in that because the experience of foul taste on sampling is immediate it involves no ambiguity as to which prey is associated with the negative experience, and it may, therefore, effectively confer protection upon untreated prey. This has been shown with a family of captive red foxes successfully conditioned to avoid untreated milk after drinking treated milk (Macdonald and Baker 2003), and may hold some potential as a predation deterrent.

Aversion and disruptive stimuli

Undesirable stimuli such as sonic and light deterrents have been tested to scare canids away from livestock and game. These can include strobes, sirens or pyrotechnics that aim to startle or frighten a predator, forcing them to retreat from an area of livestock or disrupting their predatory attempts (Shivik and Martin 2001; Shivik *et al.* 2003). Sounds are alleged to repel animals by several mechanisms including, pain, fear, communication jamming, and disorientation (Bomford and O'Brien 1990). Explosive bangs deliver sound loud enough to cause pain, but often are a nuisance to humans and the use of pain for animal control tends to draw animal welfare objections. Another disadvantage to these sound repellents is that animals quickly become habituated to them, rendering them ineffective in the long-term.

A more advanced and novel use of disruptive stimuli is to coordinate the activation of the stimuli with the actual predation behaviour. Disruptive stimuli could be triggered by collars worn by individual canids, so that the stimuli are activated on the approach of the predator toward a certain area, i.e., a pasture containing livestock (Shivik *et al.* 2003). Collars can also be fitted that will give a canid an electric shock if they attempt to attack livestock, causing pain and discomfort and hence repelling the individual (Shivik *et al.* 2002). These collars could be placed on target animals by using automated collaring devices (Rasmussen 1997; Shivik *et al.* 2000), and by placing these collaring devices in the vicinity of livestock herds it may be possible to target collaring to those animals which are most likely to attack livestock. The potential for this approach is somewhat limited due to high costs of equipment and the level of expertise needed to set up the systems. However, they may prove useful in areas where the conservation of a threatened predator is paramount, thereby justifying the high cost.

Fertility control

In theory, canid populations could be manipulated by controlling their reproductive capacity, and fertility control via immunocontraception and chemosterilants have been proposed for such population control (e.g., Asa 1992). In

Australia, immunocontraception of red foxes has been tested to deliver protection from predation to marsupials (Newsome 1995). This approach could prove useful for many common canid species involved in conflict, but current technology has the disadvantage that a high 'hit rate' is required to achieve population control, as well as the need of handling, and the ensuing high cost and effort. Baiting with birth control chemicals may become a viable option in the future, pending the development of easy to use, reliable and safe birth control compounds.

13.3.3 Approaches to increase tolerance of canids

The traditional approach to dealing with troublesome predators is giving way to a more compassionate one that also focuses on changing human perceptions and behaviour in an attempt to reduce conflict while coexisting with wild species. For this to happen, a greater awareness of the views of all the relevant stakeholders (e.g., livestock producers, wildlife managers, hunters, conservationists, the public at large), and a willingness to work together toward solutions is essential if we are to be successful at reducing conflict. By changing the attitudes of those affected and increasing the threshold of what people are prepared to tolerate, an otherwise insurmountable conflict may become manageable. In addition, economic benefits may be accrued from conserving canids and other wildlife, such as from tourism and employment, or the broader society may bear a share of the costs, through compensation and insurance schemes.

Recognising the problem

One of the first steps in reducing human-canid conflict is to acknowledge that there is a problem and to view it with objectivity (Sillero-Zubiri and Laurenson 2001). The negative impact of canids on a local economy tends to affect well-defined communities, be it small-scale shepherds in Africa or the Carpathian mountains, gamekeepers on the British moorlands, Argentine and Australian sheep farmers or Rocky Mountains cattle ranchers. Often these groups feel they are marginalised or ignored by government officials, particularly conservationists. As in many walks of life, simply listening to a grievance and recognising a community's problem can alleviate the problem through reducing underlying tension.

Community participation and sharing revenue

Novel ideas for the co-management of habitat and wildlife with local communities are increasingly seen as the way forward for conservation, particularly outside protected areas. These frequently involve improving the economic benefits the community may derive from wildlife. Community participation in wildlife management might involve the design and management of a protected area,

such as the Afroalpine grasslands of Menz that afford protection to Ethiopian wolves (*Canis simensis*) (Malcolm and Ashenafi 1997), or the actual transfer of land and resource rights to local communities. In Canada's Western Arctic, polar bears (*Ursus maritimus*) have benefited from a co-operative wildlife management process established by the Inuvialuit Land Claim, which indirectly reflects on wolf and Arctic fox conservation (Bailey *et al.* 1995). An interesting by-product of this initiative is that it overcame the mutual distrust existing between government biologists and the Inuvialuit.

There are a number of ways in which programmes have endeavoured to transfer economic benefits to local communities, particularly through ecotourism, hunting, employment and compensation for any livestock losses. Clearly, where economic benefits are substantial, this is one of the most powerful ways of reducing negative perceptions of wild carnivores and wildlife in general. The philosophy that local communities should directly benefit financially from conservation underpins many of the recent strategies for community-based conservation, although it is not without its critics.

One example of community conservation that indirectly benefited African wild dogs in a few areas is the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) in Zimbabwe. This programme was set up to promote conservation of wildlife through utilisation by allowing communal landholders to receive direct income generated by hunting fees, game-viewing and curio sales (Child 1996). Large predators, previously persecuted for livestock losses, now have enhanced value for the local people as they command substantial hunting fees. Probably of greater value, however, are conservation projects that become significant employers in the local community (e.g., the Ethiopian Wolf Conservation Programme), and may also provide additional social benefits such as supporting schools and providing healthcare.

Compensation and other cost-sharing schemes

The cost of tolerating wild canids tends to be unevenly spread, and there is a case for the broader society to share the burden with the few afflicted individuals through public funding. Compensation schemes are one such mechanism, but they have faced many limitations and may soon give way to better alternatives.

Direct compensation for livestock losses has proven to be a relatively widespread and sometimes inexpensive, but not always effective, means for relaxing opposition to canid conservation. In Italy, for example, the local government compensates 100% of the value of livestock killed by wolves, bears and even feral dogs (Cozza *et al.* 1996). This amounted to a modest 0.4–2.8% of total livestock subsidies in the region. It is vital that the criteria for compensation are clearly laid out, to avoid abuses in the claim system. For example, a few farmers may take

advantage of the situation to gain other subsidies. In Italy, farmers sometimes keep old or infirm sheep for headage payments, but these are more likely to be killed by predators (Cozza *et al.* 1996). The opposite of compensation is a bounty scheme, such as that in place with a group of Argentine sheep producers, the majority of whom get 5–10 fox bounties every year, effectively subsidising some of their losses (Novaro *et al.* 2004).

Although compensation schemes may alleviate direct losses to farmers, they do nothing to alleviate the problem, rarely deal with full costs, are open to corruption, can involve expensive bureaucracy, and tend to encourage a state of constant conflict. Furthermore, they often do not identify and improve situations where only a few farmers suffer the vast majority of losses, nor do they encourage the improvement of management systems (though these shortfalls may be alleviated if compensation criteria are modified). Crucial components of a successful compensation scheme include quick and accurate verification of damage, prompt and fair payment, sufficient and sustainable funds, and measures of success (Nyhus *et al.* 2003). As compensation schemes are costly to administer and are open to corruption, a scheme could be considered whereby rather than paying owners for each kill, they are paid a lump sum to tolerate predators. This approach would positively benefit those that have good husbandry practices and hopefully promote others to follow suit.

Some of the difficulties inherent in compensation may be circumvented by community-based insurance schemes, where the community has a vested interest in the transparency of the system and legitimacy of claims, and where producers sustaining least losses may derive some benefit analogous to a “no-claim” bonus. Commercial livestock growers may insure valuable stock against predation, particularly with pedigree herds, using established commercial insurance brokers. Premiums could then be reflected by parameters such as management strategies and risk of predation due to proximity to the wildlife area. As a result, rather than ‘managing the predator’, in order to meet regulations set by insurers, such a system would encourage ranchers to adopt an active herd management strategy (e.g., Rasmussen 1999).

In the USA, Defenders of Wildlife has created an innovative programme called the Proactive Carnivore Conservation Fund with the objectives of reducing conflicts between predators and humans, keeping predators from being unnecessarily killed by agencies in response to human conflicts, and increasing general tolerance for carnivores across the landscape. They cost-share with ranchers actions to prevent livestock depredation from occurring, such as buying livestock guardian dogs, erecting electric fencing to keep wolves away from sheep, hiring “wolf guardians” to monitor wolves in sheep territory by radio telemetry, and chasing them away when they get close to livestock (N. Fascione pers. comm.). Defenders of Wildlife has also

paid more than US\$250,000 in compensation to ranchers for losses due to wolf attacks since 1995.

Other alternatives include providing tax incentives to landowners and transferring user fees from recreation to landowners. A novel way of sharing the cost of living with carnivores is to add a premium price to goods labelled and marketed as produced by “predator-friendly” farms (e.g., Cheetah Conservation Fund, L. Marker pers. comm.), and “wolf-friendly” and “wild dog-friendly” beef would be the next natural step for this approach.

Recreational use

Ecotourism has been a major growth industry over the last 20 years and there is no doubt that some canid species are becoming an attraction for tourists planning a traditional wildlife safari. Traditionally, tourism has been a source of revenue for established conservation areas, with parts of Africa, India and the Rocky Mountains abounding with examples where safari tourism has become a major source of income. In southern Africa, commercial farmers, whose precarious income from cattle farming has always been susceptible to drought, are increasingly turning to tourism as well as consumptive trophy hunting as an alternative source of income (Lambrechts 1995).

Although ecotourism appears to be a potent tool for canid conservation, it is perhaps only the high profile and visible canid species (e.g., African wild dogs, grey wolves, Ethiopian wolves, dholes and maned wolves) that may draw tourists, and hence may be partially capable of supporting a sustainable tourist trade. This approach may be unsuitable to other more secretive species or those extremely sensitive to human pressure. However, it is not necessarily seeing the animal that is important, as many visitors to areas renowned for their predators are attracted by the knowledge of the presence of these animals, even if the chance of sightings may be minimal. But expectations may surpass reality and tourists and experienced naturalists alike may become disappointed if they do not see the elusive focus of their interest during their visit.

However, a note of caution is needed as the economic rewards of ecotourism may be low or not reach the expectations of the local community. For example, in the Bale Mountains in Ethiopia, where income from tourism is often given as a justification to the local community for the presence of a park, the number of tourists visiting each year is numbered only in the hundreds, many visitors hire vehicles rather than local horses or guides to see the area, and the amount of money that goes into the local community is relatively small. Tourism may also be susceptible to changes in the global economy or political stability of a given country. This was dramatically displayed by the recent crash in Zimbabwe’s substantial tourism trade (with African wild dogs a significant attraction), due to ongoing political instability. Thus, it would be unwise to hinge carnivore conservation purely on the economic benefits accrued from

tourism, particularly as only a proportion of these benefits may go to local communities. It is also salient to point out that many regions do not necessarily lend themselves to ecotourism, and, furthermore, many rural communities may not welcome the intrusion of paying visitors.

Conservation education

In many situations it is impossible to provide sufficient economic benefit to local communities to compensate for the resources that are lost to wildlife (see Chapter 18). In these circumstances the most important way that public support can be gained for large canids and their conservation is through educational programmes, so that local people can relate positively to the species or habitats in question. Perceptions of predator problems often exaggerate the reality (e.g., Rasmussen 1999), and education programmes can target this by delivering accurate information and increase people’s tolerance and appreciation for wildlife (Conover 2002).

Recommendations to involve the local community include targeting key groups with education campaigns, building support through the use of spokespeople within the target groups, integrating human and ecological

Signage for city residents regarding urban-dwelling coyotes as part of “Co-existing with Coyotes” programme. The programme works to reduce conflict between people, pets and coyotes through education. It targets elementary school children, day care facilities, park users and pet service businesses as audiences as well as providing situation specific advice and information to individuals who have encountered a coyote. Vancouver, British Columbia, Canada, 2003.

Coyotes have large ears that point up. They can hear a mouse under 20cm of snow.

Coyotes rarely fight with each other. They use gestures and sounds to communicate.

The majority of coyotes that have bitten children have been fed by adults.

Coyotes have bushy black tipped tails which they carry low while in motion.

Coyotes are active day and night.

Coyotes eat a wide range of foods including rodents, fruit, insects and fish.

Lower Mainland Coyotes weigh between 9-16kg (20-35lbs).

COYOTE ALERT

Think you saw a coyote? You probably did. Coyotes are very adaptable creatures and are commonly found in cities across North America. They moved into Vancouver in the late 1980s.

Pet safety:

- Keep your dog in sight and under control
- Avoid bushy areas or neglected properties
- Never let large dogs interact with a coyote
- Never feed coyotes

If a coyote approaches:

Appear as aggressive as possible

- Shout in a loud and deep voice
- Throw objects at the coyote
- Do not run, maintain eye contact
- Move towards an area of activity

Coyote attractants in your neighborhood:

- Accessible garbage, compost, fallen tree fruit
- Rat habitat - neglected sheds and properties
- Outdoor pet food (stored or fed)

Coyote Info-line: 604 681 WILD (9453)
www.stanleyparkecolgy.ca

Stanley Park Ecology Society | Report aggressive coyotes or the feeding of coyotes to the Ministry of Water, Land and Air Protection at 604 582 5200.

concerns and, if possible, designing species-specific education initiatives using the species as a flagship for other conservation concerns. Some canid species can act as such flagships to gain public support for habitat conservation – grey and Ethiopian wolves being good examples, with comparable potential for African wild dogs, dholes and perhaps even Darwin’s or island foxes. Conservation projects may engender a local pride so that the community see the target species as “their” animal, and this is more often than not the root of the problem.

In the recent past educational activities have been seen as the first step in outreach programmes around protected areas, and indeed are increasingly becoming an integral part of the activities of conservation agencies, but their success is still open to question and not often evaluated. In a few cases involving canids that we reviewed, however, there was an obvious improvement in the situation. For example, in northern Kenya the killing of wild dog pups at dens stopped (K. Doherty pers. comm.) and in Canada an education programme made farmers aware of the presence and conservation importance of swift foxes and the farmers became involved in monitoring foxes (A. Moehrenschrager pers. comm.). For education programmes to reach their full potential as formal undertakings, it is imperative that adequate funding, resources, and trained personnel are available, as opposed to being additions “tagged” onto existing research programmes.

13.4 Conclusions

Increasingly, farmers are re-examining traditional anti-predation methods and, with the use of new technology,

adapting and developing them to fit into modern livestock production systems. Simple husbandry practices, such as keeping livestock in pens at night, extra surveillance by herdsmen and shepherds, the provision of alternative pastures away from canid dens, the proper disposal of livestock carcasses, and the use of livestock guarding dogs (particularly breeds with more developed anti-predator instincts), may all help to reduce livestock losses.

In some cases, it may be difficult to change livestock husbandry, either because farmers are resistant to change or because management options are too difficult or expensive to put into practice. There may also simply be little incentive for farmers to change, if losses are relatively low, or else farmers may not be keen to take on perceived extra work. Thus, it is important that the local community is involved by any process seeking to address human-canid conflict through the various conservation approaches reviewed, in the hope of reverting traditionally, deeply based, negative views of wild canids in younger generations. Community involvement, however, requires strong partnerships, shared goals for both wildlife and human communities, and shared responsibility.

Realistically, in human-dominated landscapes where canids and people coexist there will only be, at best, an uneasy tolerance. Thus, to conserve wild canids, conservation policy must encompass a mixture of strategies, including protectionism, conservation education, public relations, community involvement and revenue sharing. While some of the examples in this review have illustrated how steps have been taken along this path, future conservation efforts must expand the use and variety of innovative and imaginative solutions to canid-human conflicts.

Evaluating and Predicting the Impacts of Exploitation and Trade on Canid Populations

W.E. Johnson

14.1 Introduction

Canid exploitation and trade forms an integral part of our cultural heritage, and, to a large extent, reflects the full range of our contradictory attitudes and responses to wildlife in general. Canids, having been exploited for subsistence, medicinal, commercial profit, and recreational purposes, have also been domesticated for the purposes of pets, companions, co-workers, and service providers. While large investments of time and money have been employed to persecute individuals and populations as pests, because of concerns over personal safety, disease transmission, or depredation on livestock and pets, substantial resources have also been spent on protecting and restoring canid populations.

The exploitation of canids often elicits strong feelings and disagreements. However, given that harvest and trade of wildlife, is not, in and of itself, inherently intolerable, it becomes easier to find commonly accepted approaches and links between the exploitation of canids and conservation. Market regulation, if structured correctly, can be easier, more effective, and less expensive than their elimination. For many species there is growing support for the concept that providing viable commercial outlets for wildlife products can be a powerful incentive for the conservation of these populations, provided that resource ownership and profit benefits are equitably distributed.

The steps needed to monitor population health and the effects of harvest or persecution, through the utilisation of increasingly sophisticated methods to examine population dynamics, predator-prey relationships, and habitat requirements, are generally agreed upon. However, the task of analysing the costs and benefits of wildlife trade is substantially more complicated. A significant issue is the difficulty (technically, economically, and politically) of obtaining sufficient data on the level of trade, and in obtaining appropriate biological and demographic data from elusive canid species. Although there have been substantial improvements in population sampling and estimation techniques (see Chapter 15), as well as an increased understanding of the parameters that are important in maintaining viable populations, for most species and populations, neither the needed baseline data have been collected, nor have populations been monitored

for sufficient periods of time to make broad conclusions about how to distinguish between detrimental and non-detrimental impacts.

14.2 The positive impacts

There are numerous examples of the positive impacts of trade in canids. Several canid populations have successfully sustained varying levels of harvest and trade for extended periods of time without long-term deleterious effects, while supporting viable economic enterprises. The red fox (*Vulpes vulpes*), gray fox (*Urocyon cinereoargenteus*), coyote (*Canis latrans*), and Arctic fox (*Alopex lagopus*) are the best examples of how exploitation over an extended period of time can be biologically and economically sustainable (Geist 1994). The long-term success and sustainability of this exploitation has been attributed to several factors, including public ownership of wildlife, elimination of a commercial market for the parts of vulnerable wildlife species, centralisation of wildlife laws, and prohibition of indiscriminate killing (Geist 1988). The sustainability of exploitation also appears to be facilitated when the commercial value of the wildlife products is low relative to alternative sources of income, and when the investment by the hunter or trapper in terms of time and money is high (Lavigne *et al.* 1996). The sustainability of exploitation can also be self fulfilling as the increased monitoring, scrutiny, and regulation that accompanies legal exploitation activities provides additional data and science that promotes better population management of these populations.

Utilisation is also more likely to be sustainable when supported by increased economic benefits derived from related activities. In North America, the Arctic fox is also a prime example of the economic benefits that canid exploitation can bring to local communities, as it remains the single most important terrestrial game species in the Arctic and an important source of income for the First Nations peoples. In addition, a large percentage of the trade in canids has historically been for fur derived from captive populations, especially for Arctic fox and raccoon dog (*Nyctereutes procyonoides*) furs farmed in Scandinavian countries and Poland.

14.3 The negative impacts

Humans can also negatively affect canid populations in a direct manner. These impacts often lead to local extirpations of species, and more rarely, to their extinction, as exemplified with the Falkland Island wolf (*Dusicyon australis*). The legal market of wildlife products, either nationally or internationally, especially when valuable, can promote overexploitation and illegal poaching. However, one of the most common factors leading to large reductions in numbers and distribution is human exploitation and predator control. For example, grey wolves (*Canis lupus*) have cohabitated with humans for thousands of years, and have invariably been extirpated through direct persecution and indirectly by the loss of habitat and prey. Due to their threat on humans and livestock, wolves were totally eradicated from large parts of North America and most of Europe by 1900 (Delibes 1990; Young and Goldman 1994) through hunting and poisoning, and were deliberately removed from all of the Japanese islands, except Sakhalin, in the 19th century (Dobson 1994). However, heavy exploitation does not always lead to extinction. In contrast with wolves, coyotes have managed to expand their numbers and distribution in spite of tremendous efforts to eradicate them. Coyotes, which benefited from the removal of wolves throughout much of North America, thrive in most human-inhabited areas where other carnivores do not.

14.4 Case studies

The primary motivations behind the wide-scale exploitation or persecution of canids have generally been commerce and population control, with intensity or effort varying predictably on supply and demand. Major differences in the commercial exploitation of canids are seen among species, populations, and administrative divisions (regions, countries, etc.), many of which are illustrative of how canids respond to commercial harvest, and how exploitation and conservation are often linked. An illustrative example of the many issues relating to canid trade and harvest is the case of three common fox species found in the southern cone of South America. Each has been persecuted to varying degrees in large portions of their ranges, primarily for economic and commercial purposes: the culpeo (*Pseudalopex culpaeus*) and the chilla (*P. griseus*) in Chile, Argentina, Bolivia, and Peru, and Pampas fox (*P. gymnocercus*) in southern Brazil, Uruguay, Paraguay, and northern Argentina. Although some of these foxes are exploited for subsistence purposes, predator control, or recreation, historically, the major factors motivating their harvest have been economic.

During the 1970s, foxes were among the most commonly traded species in South America. From 1976 to

1979, around 3,600,000 'grey' fox pelts (chilla and Pampas fox combined) and 32,000 culpeo fox pelts were exported from Buenos Aires, Argentina, with a total value of over US\$80 million (Mares and Ojeda 1984). From 1982 to 1984 these figures had dropped to around 70,000 'grey' and 4,500 culpeo fox pelts (Ojasti 1993). This decline was probably the result of a combination of reduced demand in the foreign markets of the United States, Germany, and the rest of Europe, stricter enforcement of regional wildlife legislation, and reductions in population densities in the most readily accessible areas (see sections 3.4 and 3.6).

In South America, as with elsewhere in the world, fox exploitation is also related to the trade of other species, such as wild cats (McMahan 1986; Bowles 1996), and to the dynamics of national and international markets, laws and regulations for these species, and the strength of local enforcement agencies. Trade in carnivore skins in South America focused initially on the jaguar (*Panthera onca*). Later, it shifted to foxes and smaller cats such as the ocelot (*Leopardus pardalis*), margay (*L. wiedii*), oncilla (*Oncifelis tigrina*), and eventually to the Geoffroy's cat (*O. geoffroyi*), as laws regulating exploitation and trade shifted both nationally and internationally in the 1960s and 1970s. Traders and professional trappers were motivated to develop new wildlife products for trade and to find commercial routes through countries with the least enforcement and the most relaxed laws. Since the inception of the CITES agreement, the Appendix status of several species of cats and foxes had to be changed because of unsustainable trade or the difficulty in distinguishing products from protected and unprotected species (Bowles 1996).

Although the international pelt trade has, by most accounts, continued to decline during the last two decades, trade still occurs, especially in Argentina (e.g., Iriarte and Jaksic 1986; Johnson and Franklin 1994a; Novaro 1995). However, small farmers, ranchers or herdsmen have generally replaced the professional hunts or trappers, often harvesting foxes as "by-catch" while spotlighting or trapping for other economically important species such as the European hare (*Lepus capensis*). Significant economic incentives still exist for the continued exploitation of these carnivores. The money derived from the sale of a few pelts a year can represent an important contribution to the annual income of many rural residents (Novaro 1995). These factors, along with cultural traditions, sustain trapping throughout most of southern South America, despite restrictions. Culpeo hunting, in particular, continues in an effort to reduce sheep depredation (Bruggers and Zaccagnini 1994). A limited amount of trade in fox pelts still occurs in southern South America, including a small number of farmed animals. However, exports have declined from the levels of the early and mid-1980s, probably due to lower international demand. For example, from 1997 to 1999 around 8,000 fox pelts were



Ranch worker with culpeo skins. Neuquen province, Argentina, 2000.

Andrés Novaro

exported from Argentina annually (A. Novaro and M. Funes pers. comm.).

Although the legal trade of pelts in South America has been credited with leading to overexploitation of target and non-target populations and species, the impact of harvest on the South American fox populations is unknown, mostly because there have never been any broad estimates of population sizes. Many of these fox species appear to tolerate high levels of exploitation (Broad *et al.* 1988). The ability of South American foxes, like other canid species, to tolerate such high mortality is not only due in part to their generally high intrinsic growth rate and large reproductive capacity, but may also be a function of the heterogeneous spatial distribution of hunting pressure. This in effect creates “source” populations of non-exploited foxes and “sink” populations of exploited animals (Pulliam 1988). Novaro (1995) concluded that in spite of unsustainable trapping levels on some ranches (46 to 73%), fox densities remained relatively stable because of recruitment from other areas. Although fox populations have also benefited from the establishment of National Parks and other protected areas, this does not always guarantee their protection. In Torres del Paine National Park, 45% of the documented chilla and culpeo mortality resulted from poaching (Johnson and Franklin 1994a).

Largely because of difficulties in regulating and limiting the legal hunting, trapping, and trade of foxes in South America, in many areas the most effective conservation tool for maintaining carnivore populations has been to completely ban these activities. In contrast, the characteristics and history of carnivore exploitation in North America have been somewhat different. In Canada

and the United States a combination of greater monetary resources, stronger governmental agencies, broader and stricter enforcement of hunting and trapping limits, and better population estimates have led to the management of carnivore populations through legal harvest. Hunters and trappers, as well as a large segment of the general public, are often active participants in the process of managing populations to ensure long-term viability. This has led to the maintenance of legal hunting and trapping of numerous carnivore species including foxes, cats, bears, and mustelids, as well as a wide range of ungulates and birds.

Among the northern hemisphere canids, Arctic foxes, red foxes, and raccoon dogs are the most economically important furbearers and all three are seasonally trapped, and to a lesser extent raised in captivity. The current distributions of red fox and raccoon dogs have both been influenced by human introductions. Red foxes from Britain were released into north-eastern North America during the late 1600s and early 1700s, perhaps contributing to a range expansion across much of the continent around that period (see section 5.3). Raccoon dogs were originally restricted to north-eastern Asia, but after the release of several thousand individuals into western Russia from the late 1920s through the 1950s, they are now found throughout much of northern and eastern Europe (see section 5.4).

14.5 Predicting the impacts

There is much that can be applied to canid conservation that comes from studies of their exploitation and

population biology. The impacts of exploitation on the demographics of canid populations can be complex, and are affected by numerous ecological, economic, social, and legal factors. Harvest levels tend to fluctuate greatly over time and vary considerably from one area to another. Historically, the largest impact has been on species exploited for commercial gain for their pelts, although no canid species appears to have been driven to extinction from economic trade alone. The impact of exploitation has been especially acute on species with low densities and birth rates, such as the grey wolf and African wild dog (*Lycyon pictus*), during times when they have had high economic value. Species with larger population sizes, higher birth rates, lower economic value, and/or more elusiveness have fared better. Canid species in the latter category that have been heavily exploited or persecuted species include red fox, culpeo and coyote. For some species, such as the coyote, dingo and red fox, it is clear that extraordinary efforts are required to completely eliminate the species from an area. In these species, moderate harvest rates can easily be compensated for by reproduction rate or immigration. Broad, sustained control efforts are needed to maintain reduced populations (Harris and Saunders 1993; Reynolds and Tapper 1996).

Populations respond to human-induced mortality from harvest or control in many different ways. One of the most common responses is to increase recruitment. In red foxes, increasing total reproductive output more often depends on changes in pregnancy rates, especially among juveniles, and not on changes in litter size (Pils *et al.* 1981; Allen 1984; Harris and Smith 1987). Immigration and emigration are also important mechanisms that can compensate for heavy exploitation, especially in mobile species. In heavily harvested populations, emigration may become insignificant or be delayed due to the increased availability of resources (Harris and Smith 1987). Conservation efforts to increase carnivore populations might thus benefit from actions that shorten generation times or increase reproduction rates. Carnivore populations can also respond to exploitation through compensatory mortality, where harvest simply replaces other forms of death, thereby limiting its effect on populations (Errington 1956), although some degree of additive mortality has been shown in a few carnivore species (Bailey *et al.* 1986; Clark *et al.* 1989). For population managers managing species for which additive mortality is operative, even low levels of mortality from harvest or poaching can be detrimental.

In addition to affecting population size directly, demographic changes can also result from the preferential harvest of certain sex and age classes of carnivores. Trapping of canids during certain periods of year, when the young are dispersing for example, can disproportionately affect different segments of the population. Males disperse farther and in greater numbers in many canid species than females, perhaps resulting in additional differences in genetic and

demographic patterns (see review by Waser 1996). In a study of urban red fox, 73% of male juveniles dispersed compared with 32% of females (Harris and Trehwella 1988) and over greater distances (Trehwella *et al.* 1988).

The effects of exploitation and control vary among species and responses may be hard to predict because of complex interactions among the various population parameters. For example, most forms of exploitation are spatially non-random. The heavy harvest of certain populations, areas, or habitats can affect the demographic connectivity of populations and the ability of individuals to disperse, creating or exacerbating the complexity of a metapopulation structure and increasing the probability that harvest mortality may be additive instead of compensatory. Aspects of this scenario have been studied by some authors with source and sink population models incorporating differing rates of productivity, survival, and dispersal (Pulliam 1988; Danielson 1991). This approach has not often been tested empirically; however, it has been proposed that dingo populations experience a cycle based on food supply and control. When food resources are scarce in the safe source areas, they disperse to agricultural areas where fewer dingoes and intense persecution and control efforts (e.g., poisoning, trapping, shooting) create demographic sinks (Thomson *et al.* 1992). In south-eastern Australia, this cycle has been disrupted by the maintenance of almost 6,000km of fence separating areas of intense control from areas where dingo populations are tolerated.

There are numerous examples where certain canid populations receive protection, while adjacent populations are harvested or persecuted. For example, the Pampas fox is hunted legally and controlled by a bounty system in portions of Argentina. However, in Brazil, Uruguay and Paraguay, populations are legally protected, although persecution still occurs from sheep ranchers. The grey wolf, while common in Canada, Alaska, and Russia, where the species is harvested for furs and persecuted for perceived threats against livestock, game species, and humans, is under threat elsewhere in the world. As with other species, although it is unlikely that harvest and trade of wolves in Canada, Alaska and Russia threaten those populations, the effects of legal trade on other less stable populations are not well understood.

The lack of sufficient data by which to anticipate the impact of exploitation, population control, or conservation efforts is one of the largest problems facing wildlife managers (Harris and Saunders 1993; Weber and Rabinowitz 1996). Field research is crucial to understanding the basic ecological requirements of exploited species. However, it is as important to monitor the results of management plans, not only to be able to modify these actions, but also to learn from them. To this end, long-term data sets are particularly valuable. Good data has an additional value when enacting management

plans in that some of the paralysing debates over policy decisions can sometimes be avoided when discussing facts instead of dealing with opinions. Since we will probably be making active management decisions about all carnivore populations in the future, we must accept the responsibility to do this as well as possible. This implies being able to collect the data with which to make these decisions.

To improve our ability to manage canid populations, both for harvest and conservation, research is needed in several areas. Some of the needed research efforts are species-specific while others are more broadly applicable. For many canids, basic information on life-history parameters and population ecology is still unknown, much less how they will respond to exploitation. The management of many canids would benefit from research similar to that which has been conducted on wolves, which has allowed simple models to be written estimating sustainable mortality rates, and estimating the sizes of wolf populations given available ungulate biomass (Fuller 1989). These data would then allow managers to predict the resiliency of populations to exploitation, factoring in their aptitude to alter their behavioural patterns, their capacity to compensate demographically to increased exploitation levels, and their ability to disperse across different habitats, distances, and barriers (Weaver *et al.* 1996). More research is also needed on the importance of heterogeneous harvest levels and the role of refugia, which leads to source-sink dynamics.

More research is needed comparing the effects of different exploitation techniques on different species. For example, the timing and location of harvest could theoretically influence the age and sex of individuals killed. Often only the number of animals harvested is taken into consideration when assessing whether populations are threatened. However, harvest based on reproductive potential, but which does not consider behavioural aspects might severely disrupt relationships in regard to territories or social groups, as with wolves (Haber 1996). More research is also needed comparing exploited and non-exploited populations of the same species, in similar environments.

The difficulty in obtaining reliable population estimates is one of the most fundamental tools missing for managers. Increased emphasis needs to be placed on the development of both direct and indirect methods of monitoring populations using new technologies. This should not only include demographic characteristics, but also genetic aspects such as reliable estimates of effective population size and the amount of gene flow among areas, as well as approaches that assess the prevalence of pathogens and their impact on the population. It is important to maintain the integrity of a functional ecosystem, since exploitation can change the ecological relationships in a community. For example, small carnivores have been shown to benefit from the loss of larger ones (see Johnson *et al.* 1996).

14.6 Conclusions

The canid family includes a diverse range of species, each with a unique set of conservation issues. In addition, the populations of many canid species experience a wide range of pressures. These differences lead to the necessity of developing coherent international, national, and local management plans for most canid species. To ensure the survival of some canids, even species such as the island fox (*Urocyon littoralis*), Darwin's fox (*Pseudalopex fulvipes*), Ethiopian wolf (*Canis simensis*), and African wild dog, for which there has never been a large commercial interest, complete protection and active monitoring and management may always be necessary. Species requiring constant attention will generally be found in small, isolated populations and have specialised ecological requirements and/or low intrinsic growth rates. These may or may not require large amounts of contiguous habitat, depending on their density, ecological requirements and dispersal abilities. Other canids are species that will require a certain amount of control in addition to thorough protection. They will include some of the larger species such as wolves, which in most portions of their range are unlikely to ever be common enough to permit controlled harvest, but which may require selective culling in certain populations, especially of "problem animals" (see Mech 1995b). Since predator control is also expensive, these activities will generally be restricted to agricultural areas.

A second group of canids will have very heterogeneous distributions and needs. In some areas populations may allow sustainable harvest, but in other areas exploitation leading to trade may be impractical or detrimental. These populations will generally be large and distributed over broad geographic areas, and present rapid growth rates and some economic value. Not all of these species will necessarily be harvested. However, for ecological, political or social reasons, any exploitation activity will surely require concurrent research, monitoring, and regulatory efforts. The species most often included in large economic trade enterprises include the red fox, Arctic fox, coyote, grey wolf, swift fox (*Vulpes velox*), kit fox (*V. macrotis*), gray fox, culpeo, chilla, Pampas fox, black-backed jackal (*Canis mesomelas*), Blanford's fox (*Vulpes cana*), raccoon dog, and dhole. Some of these, including the red fox, Arctic fox, coyote, grey wolf, gray fox, culpeo, chilla, Pampas fox and black-backed jackal, are killed locally primarily during predator control operations, and, like the dingo, have no large intrinsic economic value.

A final group of species is composed of those that generally are not exploited and for which there is limited commercial use or trade. This group includes the short-eared dog (*Atelocynus microtis*), Sechuran fox (*Pseudalopex sechurae*), bush dog (*Speothos venaticus*), pale fox (*Vulpes pallida*), Indian fox (*V. bengalensis*), corsac fox (*V. corsac*), Tibetan fox (*V. ferrilata*), golden jackal (*Canis aureus*),

Ethiopian wolf, maned wolf (*Chrysocyon brachyurus*), crab-eating fox (*Cerdocyon thous*), and hoary fox (*Pseudalopex vetulus*).

The conservation, harvest, exploitation and trade of canids are not always incompatible activities. The goals of minimising the potential adverse effects of carnivores, maximising the benefits that they provide, and also ensuring their long-term conservation, are all fundamentally related by the science of population management (Shea *et al.* 1998). The successful reestablishment and conservation of many carnivores will eventually require varying degrees of control of these same species. Inclusion of the local human community is one of the most crucial steps in promoting the successful maintenance or reintroduction of carnivore species. The debate over sustainable use of canids can also add a scientific framework to the discussions of wildlife management. The process of regulated exploitation and harvest may help enlist public support for conservation efforts, foster local participation in finding solutions to

conservation and management problems, and provide monetary compensation to the community for the presence of a carnivore population.

Exploitation of wild animals is an emotional issue that has been the subject of intense debate on moral, pragmatic, and economic grounds (see Robinson and Redford 1991; Swanson and Barbier 1992; Taylor and Dunston 1996). Humans affect all wildlife species and all wildlife management has animal welfare implications (Taylor and Dunston 1996). Successful wildlife utilisation and trade must be compatible with conservation, and generally has been based on broad ecosystem and cultural approaches. This includes the development of the necessary infrastructure and trained personnel for the management and preservation of all species in a multi-use, multi-species approach. Canid conservation will ultimately depend on the collective education of consumers of carnivore products (both extractive and non-extractive), resource managers, and those communities living in closest proximity to, and interacting most directly with these animals.

Survey and Census Techniques for Canids

E.M. Gese

15.1 Introduction

We already know that the status and distribution of canid populations throughout the world is of growing concern for biologists and the public alike. Habitat loss, fragmentation and degradation, human persecution, decreases in prey, disease, poaching, and increased competition with other carnivores due to reduced space and habitat, have led to some canid species facing extinction, while others occupy only a fraction of their former range. While reintroductions of some species have been successful (e.g., grey wolves *Canis lupus* to the Northern Rockies of the U.S.), other species face an uncertain future (e.g., African wild dogs *Lycaon pictus*). Paramount to canid recovery, reintroduction, or management, is acquiring accurate information regarding the status of a species, or a particular population. Reliable methods that provide accurate data on the distribution, abundance, and population trend of a species are required. These parameters are also fundamental for helping to determine the conservation status of a species according to the IUCN Red List Categories and Criteria (for example, the B criterion relies on knowledge of geographic range size, where a species with a range of less than 20,000km² could qualify in one of the categories of threat). However, because many canids are secretive, nocturnal, wide ranging, in densely vegetated habitats or remote areas, or at extremely low densities, surveys of a canid species or population can be very difficult.

Abundance may be assessed in two ways: relative and absolute. Relative abundance uses indices of animal abundance (e.g., track counts, dens) that can be compared over time or between areas. Absolute abundance involves actually counting animals and estimating the number or density of animals in the population. With repeated sampling over time, both relative indices and absolute estimates can be used to monitor population trends. This chapter reviews techniques useful for censusing canids, and is adapted from Gese (2001). For techniques related to determining demographic parameters (birth, death, emigration, immigration), readers are referred to Caughley (1977), White and Garrott (1990), Royama (1992), and Thompson *et al.* (1998). Methods for censusing or surveying wild canids vary in accuracy, reliability and cost. Many of the techniques described herein require in-depth evaluation as to their accuracy and reliability in monitoring population trends (Gese 2001). As an example, a recent

study by Schauster *et al.* (2002a) compared six survey techniques for monitoring abundance of swift foxes (*Vulpes velox*) in Colorado, USA. This study found that mark-recapture estimates ($r = 0.711$) were the best predictor of fox density, followed by scat deposition surveys ($r = 0.697$), scent-post surveys ($r = 0.608$), spotlight surveys ($r = 0.420$), trapping surveys ($r = 0.326$), and lastly, activity index surveys ($r = 0.067$). Combinations of techniques increased prediction capabilities. Other studies that used, or attempted to use, the techniques described in this chapter have been included as examples.

Some considerations before implementing a survey

Prior to surveying any canid population, the precision, accuracy, power, sample size, survey design, and statistical assumptions of each method should be considered (Skalski and Robson 1992). In addition, for each method the observer must address problems pertaining to “observability” or “catchability” of the species, the size of area to be sampled, costs, logistics, manpower, and time constraints (Lancia *et al.* 1994).

15.2 Methods employed to determine species distribution

Sometimes it may only be necessary to determine the presence and distribution of a species. Methods typically used to determine species distribution include habitat mapping, questionnaires, interviews, sighting reports, or confirmation of sign. Any survey method that provides an estimate of animal abundance provides distribution information as well.

15.2.1 Habitat mapping

Time can be saved by considering the type of habitat required for a species and examination of habitat maps or aerial photos. Habitat suitability models have been developed for many wildlife species (e.g., Boyle and Fendley 1987; Rogers and Allen 1987), but have not been developed for canids. With the continued development of satellite imagery, remote sensing, and Geographic Information Systems (GIS), areas containing suitable habitat for a species can be identified allowing for maximisation of survey efforts. Surveys can then be stratified by habitat types or land classes (Macdonald *et al.* 1998).

15.2.2 Questionnaires, interviews, and sighting reports

Many agencies compile status reports using questionnaires to assess the relative abundance and distribution of canid species. Sightings and general impressions from people in the field can determine species distribution, and gain a subjective estimate of animal abundance. More in-depth questionnaires or interviews of persons with knowledge of the area and who spend considerable time in the field provide not only a range report, but may also provide an estimate of abundance (e.g., Allen and Sargeant 1975; Harris 1981). Questionnaires, interviews, and sighting reports have been used to determine distribution, and sometimes abundance of several species (e.g., Allen and Sargeant 1975; Harris 1981; Fuller *et al.* 1992; Fanshawe *et al.* 1997). Problems with this method include misidentification of species, low response levels to the questionnaire, a bias for animal sightings concentrated along roads or near human habitation, and the reliability of the respondents.

15.2.3 Presence of sign

In the absence of visual confirmation of the species itself, surveys of animal sign may be used to determine presence. Several different methods of sign surveys have been used, including documentation of tracks, scats, scratches, burrows or dens, and hair samples (often obtained through the use of hair snares or hair tubes). The use of track plates to determine species presence has proven useful (e.g., Zielinski and Truex 1995). A full description is provided by Zielinski (1995), but track surfaces may generally be produced from smoked or carbon-sooted aluminum plates, contact paper, chalk, or ink. A visual and/or olfactory lure is used as an attractant and while investigating the attractant, the animal leaves tracks on the tracking surface. Identification of tracks, getting the animal to step on the plate, transportation of the plates, and protecting the track plates from weather are just a few of the common problems that require prior planning (Zielinski 1995; Zielinski and Truex 1995). This technique provides a reliable measure of species distribution or presence, but may be unreliable for determining relative animal abundance.

A common problem with using sign to determine canid distribution is the consistent identification of tracks, scats, burrows, and hair samples. Species identification from scats can be facilitated by using faecal bile acid patterns (e.g., Major *et al.* 1980). Examination of hair samples with a light microscope and comparison to a hair key or reference collection can aid species identification (e.g., Adorjan and Kolenosky 1969; Moore *et al.* 1974). DNA techniques allow for more accurate identification from scat or hair samples (Foran *et al.* 1997a,b; Paxinos *et al.* 1997), and

can also be used to identify individual animals allowing for estimation of population size (e.g., Kohn *et al.* 1999). When using scat surveys, the seasonal decay rate of the scats may need to be considered, as well as whether scats are being consumed by scavengers. Also, the amount of sign left behind by an animal does not always correlate with animal density, nor does failure to find sign necessarily indicate species absence.

In their most rudimentary form, sign surveys provide distribution information. When standardised, these sign surveys may be used as an index of animal abundance. If certain areas or habitats are repeatedly surveyed over time and the number of hours of searching (or some measure of effort) is recorded, then surveys may be standardised to allow for trends over time or comparisons between areas.

15.2.4 Remote cameras

The use of remote cameras set along trails, near bait stations, or nests has been used mainly to detect forest carnivores. The cameras, commercially available from several manufacturers (Kucera *et al.* 1995), can be triggered by an animal tripping a line, or activated remotely by pressure-sensitive plates, motion or heat detectors, or breaking of an infrared beam.

15.3 Methods for estimating animal abundance

After determining species distribution, data on animal abundance and population trends may be required. Animal abundance may be monitored indirectly by counting animal sign, or by direct methods of counting the animals themselves. Estimating animal abundance requires consistent and standardised application of a technique to be able to detect changes or differences with some degree of accuracy, precision, and power. Therefore, for the following techniques one must maintain a standardised protocol for the survey and consistently apply it to all future surveys. Whether sign surveys, indices of relative abundance, or measures of absolute animal abundance are used, caution should be exercised when examining population trends. Assessing rates of increase or decrease from trend data should take into account the precision and accuracy of the methods used. The influence of other variables on survey results should also be taken into consideration, such as characteristics of the animals themselves, topography and vegetation, temporal factors, observer experience, ability, and fatigue, and spatial distribution of the species. One should examine the assumptions and power of the technique to determine its ability to detect population changes (Gerrodette 1987; Eberhardt and Simmons 1992).

15.3.1 Indirect methods

Scent-station surveys

One of the most common sign surveys used for indexing canid abundance is scent-post or scent-station surveys (Linhart and Knowlton 1975; Roughton and Sweeny 1982; Schauster *et al.* 2002a). Scent-station surveys involve placing a scented tablet or other attractant within a circular area of sifted dirt. Tracks left by an animal are identified and recorded. Typically, stations are spaced at predetermined intervals along roads or trails and then visited for 3–4 consecutive nights to record tracks; the sifted area is swept smooth after each night. The frequency of animal visitation to operable stations (i.e., those not disturbed by wind, rain, vehicles) is used as an index of abundance. Scent-post surveys have been used to estimate the relative abundance of many canid species (e.g., Linhart and Knowlton 1975; Travaini *et al.* 1996; Sargeant *et al.* 1998; Schauster *et al.* 2002a). Seasonal changes in habitat use and visits to multiple stations by a single animal can contribute to invalid correlations of animal density and visitation rates; see Smith *et al.* (1994) and Sargeant *et al.* (1998) for recommendations on how to use these methods appropriately. Misidentification of tracks, weather (wind, precipitation), wariness of animals, and manpower should also be considered with scent-station surveys.

Activity index

A variation of the scent-station survey that has been used to index dingo populations is the activity index (Allen and Engeman 1995; Allen *et al.* 1996). This index of animal visitation uses a sifted dirt area on a road without any scent or lure to attract animals (Schauster *et al.* 2002a). The number of track sets crossing the sifted area is used to assess relative abundance and calculate a variance estimate (Engeman *et al.* 1998).

Scat deposition transects

The rate at which scats are deposited along established roadways or trails has been used to estimate relative abundance of canids (e.g., Andelt and Andelt 1984; Crête and Messier 1987; Beltrán *et al.* 1991; Schauster *et al.* 2002a). This method involves designating transects or routes along a roadway or trail, clearing all scats from the road, then returning and collecting all scats encountered two weeks later. If transects vary in length, or the time between collections varies, then the index can be standardised to scats/km/day. A study by Knowlton (1984) found that scat deposition rates for coyotes were correlated with estimates of animal density derived from mark-recapture techniques using radio-isotope tagging of faeces. For long-term monitoring, scat transects should be conducted along the same routes at the same time of year to avoid introducing biases associated with differential prey digestibility and seasonal changes in food items

consumed (Andelt and Andelt 1984). Misidentification of scats and heavy vehicle traffic on roadways can be problematic when using scat counts. Use of DNA techniques for identifying species from scats may alleviate the problems of misidentification (Foran *et al.* 1997a,b) and identification of individual animals collected during scat deposition transects could be used to estimate population size (Paxinos *et al.* 1997; Kohn *et al.* 1999). A recent study by Harrison *et al.* (2002) compared survey techniques for estimating relative and absolute abundances of swift foxes in New Mexico. This study found that for relative abundance surveys, the most efficient technique was collection of scats followed by verification of species depositing scats with DNA analysis, while for absolute abundance surveys, trapping and re-sighting with remote cameras at bait stations was more accurate than counting unique microsatellite DNA genotypes from collected scats.

Track counts along a transect

Tracks left by canids along river beds, dry washes, sandy fire breaks or roads, or on snow-covered roads and trails have been used as a relatively simple, efficient, and inexpensive measure of relative abundance for canids (e.g., Crête and Messier 1987; Servin *et al.* 1987). Canids which occupy regions that receive snow can be monitored by counting tracks along established transects one to two days after fresh snowfall. Some pitfalls when attempting transect counts of tracks should be noted. Misidentification of tracks and low power to detect population changes can occur with track counts (Ballard *et al.* 1995). Precision can be increased by increasing sampling effort, or increasing the length of transects if censusing highly nomadic species. Much of the power of this estimator is dependent upon a high rate of encountering sign along the transects (Kendall *et al.* 1992). When working in areas with snowfall, one must also consider the condition, consistency and depth of the snow, ambient temperature, and the time of year. As is typical for any survey technique involving sign, observer experience at interpreting tracks is also crucial for consistent and reliable monitoring.

Den and burrow surveys

Ground and aerial surveys for active dens have been conducted along transects to index relative abundance of some canids, mainly foxes (e.g., Trautman *et al.* 1974; Garrott *et al.* 1983; Hersteinsson *et al.* 2000). The key to this survey technique is relatively open habitat with little vegetative cover and a species that makes conspicuous dens or burrows. These surveys can be relatively expensive (aerial searches) and/or labor intensive (ground searches). The presence of faeces or tracks at the burrow or den can assist in species identification. Ground surveys along transects can also be used to calculate the density of dens if the perpendicular distance from the transect to the den

is recorded. This technique does not work well for indexing canids with large social units. For animals that exist in packs, the number of active dens would more likely indicate the number of social units present across an area, but not the size of the social unit.

Vocalisation response surveys

For canids that utilise howls to communicate, the response rate to simulated vocalisations has been used as an index of relative abundance (e.g., Wenger and Cringan 1978; Okoniewski and Chambers 1984; Fuller and Sampson 1988; Robbins and McCreery 2003). Howling surveys typically employ recorded vocalisations, although human imitation can be used. Travelling along roads or trails and stopping at predetermined intervals, howls are produced and then observers listen for a specified amount of time for a response from the target species. A recent study using both playbacks and human simulations of long distance calls of African wild dogs recorded that dogs would approach from distances of as much as 2km, and found that playbacks are an effective conservation tool particularly where road networks are limited and/or thick vegetation restrict off-road driving (Robbins and McCreery 2003). Surveys may be conducted over several nights using the vocalisation response to estimate relative abundance. Standardisation and consistency of this method is needed for reliable and comparable results for trend analyses. The seasonal, social, temporal, and spatial factors that influence vocalisation rates also need to be noted (Harrington and Mech 1982; Walsh and Inglis 1989; Gese and Ruff 1998). For an accurate population census, the area of interest needs to be intensively surveyed to obtain adequate coverage (Fuller and Sampson 1988).

Frequency of depredation complaints

The frequency of livestock depredation complaints may be useful as an indicator of relative abundance under the general belief that animal abundance is correlated with rates of livestock predation. Because this relationship has not been explicitly tested, caution should be exercised when using this technique as depredation rates are subject to changes in livestock stocking rates, habitat type, size of area used, husbandry practices, and environmental variables (Knowlton *et al.* 1999).

Some considerations when using indirect methods

Indirect methods provide only relative abundance and must be applied consistently for any reliable comparisons between areas, habitats, or time. Whenever indices of relative abundance are used, it should be determined whether relative indices and absolute abundance are positively and linearly related. Comparison of an inexpensive indirect method to a more expensive direct method could prove worthwhile for calibration of the less expensive technique. During calibration, the techniques

should be performed concurrently and conducted on a species-specific, habitat-specific, and seasonal basis. Unfortunately, few indices of relative abundance have been tested with a known population estimate.

15.3.2 Direct counts

Direct counts involve actually counting the animals themselves, in contrast to counting sign. These counts may use either dead animals (e.g., harvest reports, mortality samples) or live animals (e.g., trapping, sightings). The assumptions of direct counts and the estimators used to determine population size should be reviewed (Caughley 1977; Burnham *et al.* 1980; Skalski and Robson 1992). Counts may involve total counts of the area, or a subsample of the area with extrapolation to the rest of the area of concern. Stratification of subsamples by habitat type can increase the validity, usefulness, and precision of the survey (Macdonald *et al.* 1998).

Harvest reports and pelt registration

One method of estimating abundance (and distribution) of a species is using historical and current harvest or trapping records (e.g., Clark and Andrews 1982). In the Canadian provinces, mandatory pelt sealing reports have been used to estimate furbearer population trends (Novak 1987). While information from harvested animals can be used to construct models for population estimation (Clark and Andrews 1982), harvest data alone is generally not a reliable estimate of population trends. Pelt prices, trapper behaviour and memory recall, differential harvest methods, and environmental and social factors all influence harvest rates (Clark and Andrews 1982). For rare species, harvest reports are generally unreliable for population trends, while harvest reports for abundant furbearers may be reliable measures of population trend. However, little in-depth testing has been conducted to confirm the relationship between population density and harvest statistics.

Road mortality samples

The frequency of carcasses found on roads has been proposed as a measure of population trend, usually as an index of relative abundance (e.g., Clark and Andrews 1982). However, differences in animal behavior and movements, habitat, traffic density, road surface, and road density likely influence kill rates of some canids. The relationship between population density and road kill rate also has not been adequately examined. Road mortality samples can confirm species presence.

Drive counts

In certain habitats, animals may be driven into an area and counted as they cross the observer's line (e.g., Beltrán *et al.* 1991). This technique is labour intensive, due to the use of

counters, beaters, and possibly hounds, and sample sizes may be difficult for statistical analyses and comparison.

Spotlight surveys

Spotlight surveys are a cost effective method typically used for assessing the relative abundance of nocturnal canids (e.g., Ralls and Eberhardt 1997; Schauster *et al.* 2002a). These surveys involve two observers standing in the back of a truck driven slowly along roadways, scanning the road and sides using spotlights. When an animal is detected, usually by eye shine, the driver stops and the observers identify the animal (sometimes using binoculars or a spotting scope). The mileage and time of detection is recorded for each sighting. An index of animals/km is then calculated. Spotlight counts can be used to estimate population size with line-transect methodology if the perpendicular distance to the sighted animal is recorded (Thompson *et al.* 1998). Transects need to be fairly lengthy, and because vegetative cover and topography influences visibility, these variables should be considered in survey design (Ralls and Eberhardt 1997). Surveys can be conducted over several nights (repeated counts) to obtain a measure of sampling error. Large samples with replication are needed to detect changes in population size with any statistical power (Ralls and Eberhardt 1997). Surveys can be conducted seasonally and annually for population trend analysis (Schauster *et al.* 2002a). Spotlight counts do not work well in areas with low densities of canids. A recent study by Ruelle *et al.* (2003) has noted that a number of methodological improvements are necessary before spotlight distance sampling can become a routine monitoring tool for fox species.

Remote camera traps

While camera systems have been used to detect species presence and identify animals at bait stations or nests, they can also be used to determine abundance if individuals can be identified by artificial tags (e.g., ear tags, radio collars) or natural features (pelage, etc.) and then apply mark-recapture estimators. Harrison *et al.* (2002) found that re-sighting with cameras at bait stations was more accurate for estimating swift fox abundance than counting unique microsatellite DNA genotypes from collected scats. Remote cameras also provide a permanent photographic record. Disadvantages of remote cameras include their expense (although the technology is becoming increasingly affordable), getting animals to trigger the camera, non-target species activating the camera, and the delay between photo acquisition and development (although digital cameras may negate this concern).

Catch-per-unit-effort

Live-trapping gives a positive confirmation of species presence (distribution) and the number of animals captured per trap night can also be used as an index of

relative abundance (e.g., Knowlton 1984; Crooks 1994; Schauster *et al.* 2002a). Trapping is expensive and labour intensive, and can be ineffective in areas with low density. In addition, standardisation of capture procedures and variation among individual trappers can cause problems.

Capture-mark-recapture

While mark-recapture is fairly time consuming, labour intensive, and costly, it has proved useful for estimating population size in canids (e.g., Roemer *et al.* 1994; Hein and Andelt 1995; Schauster *et al.* 2002a). Mark-recapture can provide relatively accurate estimates of population size if sample sizes are adequate, collection techniques are unbiased, and the basic assumptions for the population estimator are not violated (see Caughley 1977; Thompson *et al.* 1998, and references therein). This method involves capturing and marking individuals, then recapturing a number of the marked individuals again and estimating population size based upon the ratio of marked to unmarked animals recaptured using one of several models (e.g., Pollock 1981; Seber 1982). Marks employed to tag the animal include ear tags, radio collars, dyes, and physiological markers such as radioactive isotopes (Kruuk *et al.* 1980), iophenoxic acid (Knowlton *et al.* 1988), or chlorinated benzenes (Johnston *et al.* 1998). Recapture may involve physical recapture, re-sighting or photographs, returns from trappers or hunters, recapture via fecal analysis for a physiological marker, faecal DNA analysis, or a combination of these. If the extent of the area of interest is known, density estimates can be derived. Several models for population estimation (e.g., the Petersen, Jolly-Seber, and Schnabel asymptotic methods) can be used to calculate population size (Caughley 1977; Jolly 1982; Seber 1982; Thompson *et al.* 1998). Many of these models are available on computer software, such as CAPTURE (White *et al.* 1982), NOREMARK (White 1996), and EAGLES (Arnason *et al.* 1991).

Direct counts by removal

For some species that are considered pests, the removal method has been used to estimate animal abundance (e.g., Skalski *et al.* 1984). Disadvantages of this technique is the lack of knowledge of what proportion of the population was missed or not captured, and how large an area was affected by the removal. Due to the economic importance of the species, intrinsic values, and/or the social and ethical ramifications, the removal method is rarely employed.

Transect, strip, or area sampling

In certain circumstances, it may be possible to count the number of animals along transects, strips, in quadrants, or within a defined area and estimate animal population size or density (e.g., Burnham *et al.* 1980). Trends in

relative abundance can be compared from direct counts; absolute abundance may be estimated if correction factors can account for problems with 'sightability'. Population estimates can also be calculated by distance methods along line-transects (Burnham *et al.* 1980). Software programs that estimate population size using distance data along transects include DISTANCE (Buckland *et al.* 1993, Laake *et al.* 1993) and TRANSECT (Burnham *et al.* 1980). Aerial surveys typically require a large species occupying sparsely vegetated habitat allowing for maximum 'sightability'. The number of animals sighted can be affected by animal behaviour, weather, vegetation, visibility, and observer experience and fatigue. The use of ultraviolet, infrared, or thermal imagery photography may enhance "sightability" (e.g., Havens and Sharp 1998). Ground surveys are practical for animals readily viewed in open habitats. In certain situations, the entire area of interest may be surveyed, and through repeated sampling, the entire population may be counted. However, the ability to count all individuals in an area is rare, but correction factors from a radio-marked sample allow determination of a more accurate estimate of population size. For transect and sighting surveys, it is important that the different habitats within the area be sampled, not just the areas with good visibility.

Identification of individual animals

While the opportunity to directly observe canids may be considered rare, there are certain species living in national parks or reserves with open habitats that allow for direct observation and identification of all individuals in the study area. Maddock and Mills (1993) censused African wild dogs by collecting photographs from tourists and other field personnel. They were able to identify 357 wild dogs from 26 packs by examining more than 5,000 photographs. Studies using identification of individuals are usually conducted in relatively open habitat and with a species that is observable and tolerant of human presence. Also, the animals do not necessarily need to be marked for individual identification, as individuals may be re-sighted and identified indirectly. Track characteristics have been used in which tracks of individual animals were separated on the basis of characteristics and location. The main advantage of using characteristics of individual tracks for identification is that it entails less effort than a large-scale trapping programme, although the accuracy of this method in relation to changes in population size remains untested. While individual identification allows for a relatively complete count, the time and effort necessary means that this method is useful only in particular situations and is often conducted in conjunction with behaviour studies (e.g., Gese *et al.* 1996c). Again, the use of hair snares to acquire hair samples can be used with DNA sequencing to identify individuals in the population.

Radio-telemetry

The advent of radiotelemetry increased the ability to monitor secretive canids. Using this method, one can estimate the home range or territory size of an animal. It is now widely accepted that combining territory size (and overlap) with the number of members of the social unit, plus the percentage of radio-collared transients sampled from the population, density estimates can be derived for the population (e.g., Mech 1973; Fuller 1989). For more solitary species, estimates of home-range size, the extent of inter- and intrasexual home-range overlap, and the proportion of transients in the population are used to estimate population density. While radiotelemetry is labour intensive and costly, this technique provides one of the best and most reliable estimates of population density for many species. With the advent of satellite and GPS technology, more intensive monitoring of large and medium-sized canids will be possible (e.g., Ballard *et al.* 1998; Merrill *et al.* 1998), but the technology is still somewhat expensive and systems for smaller species will require further technological development.

Águas Emendadas Ecological Station is one of the most important regions to conservation in Distrito Federal, Brazil, but is threatened by urban expansion. It is a protected area, devoted solely for preservation purposes, and is home to many ecologically important native species being monitored, including the maned wolf (*Chrysocyon brachyurus*), crab-eating fox (*Cerdocyon thous*) and hoary fox (*Pseudalopex vetulus*). This radio-collared adult female maned wolf vocalises when a researcher approaches her cub. Águas Emendadas Ecological Station, Distrito Federal, Brazil, 1997.



Flávio Rodrigues

15.4 Conclusions

The methods and techniques available for determining the presence and abundance of canid species are varied, and this chapter has attempted to illustrate by means of examples some of the instances where these techniques have been applied to studies on canid populations (or other similar large predators) and the advantages and disadvantages of each. While a combination of methods

is always likely to provide the best results (see, for example, Schauster *et al.* 2002a), the feasibility and application of the appropriate methodology will always depend on factors such as the species, habitat, costs, manpower, time constraints (Lancia *et al.* 1994), and also on the kind of questions that are being addressed and the consequent accuracy and power of the statistical assumptions of each method (Skalski and Robson 1992).

Captive Canid Conservation

K.L. Bauman, C.S. Asa, J. Grisham and W. Verberkmoes

16.1 History of canids in captivity

Wild animals have been kept in captivity for thousands of years. Military conquests and trade with foreign lands created large royal collections of exotic animals in countries including China, Egypt, and England. In the 18th and 19th centuries, several large European collections were opened to the public for the first time. These menageries or zoological parks became places of public entertainment where strange and unusual animals could be seen. The Austrian Royal collection became the Tiergarten Schonbrunn (1752) and the British Royal family formalised their menagerie as the Zoological Society of London in 1828. This increase in popularity in the 19th century resulted in the opening of hundreds of new zoological parks worldwide, including the Royal Melbourne Zoological Gardens (1857), Zoological Society of Philadelphia (1874) and Jardín Zoológico Municipal de Buenos Aires (1874).

Many early zoological collections included wolves and foxes, well known to the visitors from legend and livestock predation. The Zoological Society of London (ZSL) held grey wolves (*Canis lupus*), coyotes (*C. latrans*) and raccoon dogs (*Nyctereutes procyonoides*) prior to 1900 (Crandall 1964), while the Philadelphia Zoo received their first fennec fox (*Vulpes zerda*) in 1900 (Bauman 2002). Most

canid species proved to be quite adaptable to captivity and longevity was high. For example, a grey wolf captured as a pup lived for more than 15 years at the Bronx Zoo (Crandall 1964). Many canid species reproduced quite readily, including the raccoon dog (ZSL 1877), grey wolf (Bronx Zoo 1902, and ZSL 1903), coyote (Bronx Zoo 1900), and dhole (*Cuon alpinus*) (ZSL, late 19th century) (Crandall 1964). Captive breeding success came later for some of the more sensitive canid species with the birth of the first litter of African wild dogs (*Lycan pictus*) in 1942 (M. Quick pers. comm.), fennec foxes in 1954 (Bauman 2002) and maned wolves (*Chrysocyon brachyurus*) in 1967 (Crandall 1964).

Despite these strides in captive breeding, overall success was still limited and unable to meet the demand for animals, resulting in the continuing capture of wild animals to fill exhibits. However, by the mid-20th century it had become clear that nature did not have an inexhaustible supply of animals, and the days of easy importation of wildlife for zoos were rapidly diminishing. This knowledge, combined with the increasing awareness of the responsibilities of zoos for managing wildlife, made clear the importance of accurate record keeping. Studbooks (or pedigrees) became prevalent in the late 1960s and early 1970s, and the International Species Information System (ISIS) was founded in 1973.



Captive bred African wild dog pups, born at the Mountain View Conservation and Breeding Society facility. Fort Langley, British Columbia, Canada, 2003.

Michelle Nelson

Zoos began to take a more active role in species conservation in the early 1970s with the passage of the U.S. Endangered Species Act and establishment of the Convention on the International Trade of Endangered Species of Flora and Fauna (CITES). The first opportunity for zoos to directly participate with the recovery of a canid species came when the United States listed the red wolf (*Canis rufus*) as endangered in 1967. The passage of the Endangered Species Act resulted in the creation of the United States Fish and Wildlife Service's (USFWS) Red Wolf Recovery Program, which listed captive breeding as a priority (Parker 1988). The Point Defiance Zoo was selected to develop the captive breeding programme, which included providing the space and expertise to develop the resources critical for the reintroduction programme. In 1981, the continued decline of wildlife populations led the American Zoo and Aquarium Association (AZA) to focus captive breeding efforts and to form the Species Survival Plan (SSP7) programme. The SSP7 concept was based on long-term management of a species, wherein all animals in AZA accredited zoos form a large, cooperative breeding programme. The red wolf was one of the first AZA SSP7s (1984), with 63 wolves in four zoos. In other regions of the world, conservation programmes similar to SSP7s were developed. In Europe, the European Endangered Species Programme (EEP) developed captive conservation programmes for carnivore species, while Australia, Japan, China, South Africa, and Central and South America developed similar programmes.

Established in the 1990s, Taxon Advisory Groups (TAGs) became responsible for coordinating the captive efforts for the entire taxonomic group by developing a Regional Collection Plan. The TAG also facilitates captive conservation efforts through a network of field researchers associated with universities, conservation, and governmental agencies. Data from the IUCN Canid Specialist Group are now routinely incorporated into collection management decisions. For example, the publication of the 2002–2005 AZA Canid and Hyaenid TAG Regional Collection Plan was timed to take advantage of the recommendations from the Canid Biology and Conservation Conference hosted by the Canid Specialist Group in Oxford, UK, in September 2001. Modern day zoos and zoo-based programmes for canid species provide support for canid conservation in a variety of ways including captive breeding, education programmes, research and the funding of field initiatives.

16.2 Contributions of captive canids to conservation

The history of canids in captivity provides an excellent framework from which the evolution of zoos can be seen. The contributions captive canids have made to

conservation are often overlooked, while the benefit of captive breeding continues to be debated among some conservationist biologists. For example, without captive space and expertise, neither the red wolf nor the Mexican wolf (*Canis lupus baileyi*) recovery programmes would have had the resources necessary for their reintroduction and educational programmes. Additionally, research on captive canids in the areas of reproductive physiology, genetics, veterinary medicine, nutrition and behaviour have increased our understanding of canid biology.

16.2.1 Genetic reservoirs

Captive breeding is the basic function of every zoo. This requires an understanding of the species' natural history, appropriate husbandry, proper animal health and diet, plus skilled staff, an accurate pedigree, and the space to house multiple individuals. The decisions of which animals to breed, with whom, and how often are pivotal to every captive breeding programme (Ballou and Foose 1996). It takes many years and, in most cases, many institutions working cooperatively to create a viable captive population numbering in the hundreds. In the extreme, captivity is the last refuge for a species, as has been the case for the Mexican wolf, red wolf and island fox (*Urocyon littoralis*). Ideally, captive breeding is utilised as an *ex situ* conservation tool in a proactive rather than reactive manner, and preferably always in conjunction with *in situ* conservation actions.

Zoos are often criticised for having breeding programmes for non-threatened species, yet there are justifications for this approach. If captive breeding is done properly, managing for genetic diversity, then these populations serve as genetic reservoirs (Ryder and Fleischer 1996), if needed, for reintroduction, re-stocking or genetic exchange. Clearly, not all species will require these techniques, yet the status of wild populations are often tenuous, and unforeseen stochastic events have made many species that were common not too long ago, threatened today. For example, the wild population of maned wolves, which was previously considered stable, has shown recent evidence of decline and field research has begun to determine the scope of the problem. The captive population of maned wolves represents a source of animals if needed. Similarly, the fennec fox currently is listed as Data Deficient on the IUCN Red list (Appendix 1), as are many other fox species. Since the status of these wild populations is completely unknown, it is possible that the captive population of fennec foxes may be needed in the future as a source of unrelated individuals. Additionally, the husbandry techniques utilised for captive breeding of fennec, swift (*Vulpes velox*) and island foxes may someday prove a valuable tool for saving other fox species.

16.2.2 Educational programmes

All conservation organisations work to raise public awareness of the issues facing wildlife conservation through various communication channels, such as magazine articles, posters and television documentaries. Zoos have the advantage of being able to provide people a direct connection with animals. The opportunity to see a Mexican wolf pup's antics or touch a fennec fox while learning about its desert home provides a lasting connection. Collectively, AZA zoos in the USA receive over 130 million visitors per year, more than professional baseball and football games combined. This provides a tremendous opportunity to teach visitors about the importance of the conservation of wild dogs, wolves, foxes and jackals. Many canids need the good public relations of educational programmes to bring attention to their conservation needs. Often portrayed as ravenous predators, little would be known about canids and their conservation dilemma if it were not for zoos helping spread the message to countless visitors.

Zoos also have begun to educate visitors about local conservation issues. The Santa Barbara Zoo in California has taken a leadership role in educating the public about the island fox. They are one of only six zoos in the U.S. currently exhibiting island foxes. Graphical displays explain the reasons for the decline, what biologists are doing to address the problems and actions that can be taken to save the species (for example, not bringing unvaccinated domestic dogs to the Channel Islands). They participate in the Island Fox Recovery Team run by the U.S. National Park Service, lending staff and equipment to the project. A similar programme has been proposed for swift fox. Since all captive swift fox in AZA zoos are within the native range, the educational impact on the local level could be very high. However, not all education programmes are directed at zoo visitors; the need for educational efforts in range countries is also important. The Maned Wolf SSP7 has worked with Brazilian biologists to produce and distribute a poster about maned wolves to local villages. Written in Portuguese and Spanish, the poster not only provides species information, but also strives to inspire national pride.

16.2.3 Reintroduction*

Reintroduction attempts to establish a species in an area which was once part of its historical range, but from which the species has been extirpated (Kleiman and Beck 1994). It has been well accepted that reintroductions are complex undertakings with a multitude of biological, ecological and social factors that require long-term commitment of resources. Reintroductions of canids have occurred using both wild-born (e.g., grey wolves in Yellowstone) and captive-born animals (e.g., red and Mexican wolves). There

was a brief period in the late part of the last century when zoos billed themselves as 'modern arks', a concept that promoted the idea that animals bred in captivity existed as sources for reintroduction programmes. It is now recognised that providing animals for reintroduction is merely one of many roles of a modern zoo. A study examining the use of captive-bred animals (all taxa) as sources for reintroduction suggested these releases were successful 11% of the time (Beck *et al.* 1994). Reintroduction programmes for canids have had success using captive-born individuals in red (Waddell 1996) and Mexican wolves (Lindsey and Siminski 2003), African wild dogs (Mills 1999) and swift foxes (Boitani *et al.* 2004). In African wild dogs, released packs were comprised of both wild- and captive-born individuals, whereas the red and Mexican wolf recovery programmes released only captive-bred animals. The swift fox programme utilised both captive-bred and translocated animals and did find a higher survival rate in the translocated individuals (Boitani *et al.* 2004).

Zoos have greatly contributed to the success recovery programmes, including captive breeding and reintroduction of red and Mexican wolves. These programmes have benefited from the partnership between the zoos and USFWS by building on the respective strengths of each partner. In addition, captive breeding forms the basis of island fox recovery, for which reintroduction is the ultimate goal.

Red wolf

Declared endangered by the USA in 1967, the red wolf officially became extinct in the wild in 1980 after eight years of intense trapping effort by the USFWS (Parker 1988). The decision to remove all red wolves from the wild, while aggressive, was justified by the goals of the Recovery Programme. The goal was to establish a captive breeding colony of genetically pure red wolves, as determined by genetic testing, and supply animals for the release programme (Parker 1988). The breeding colony was established through a cooperative agreement between USFWS and the Point Defiance Zoo. Forty wolves were moved to the Point Defiance Zoo for breeding and further genetic testing, and of those 17 were certified as 'pure' (Parker 1988; Bergman 1997). Infanticide occurred in the first several litters, but husbandry modifications resulted in successful reproduction from 14 of the certified wolves (Bergman 1997).

In 1984, when the red wolf programme became an SSP7, the population had grown to 63 animals at five zoos (Waddell 1995); in 2003 there were 153 animals living in 35 institutions in addition to those in the wild (Waddell 1995, 1996). Since the release of four captive pairs of red wolves in eastern North Carolina in September 1987, red wolves have formed family groups, established territories and

* Note: Additional information relating to reintroduction programmes for canids can be found in Chapter 17.

produced young. Since the beginning of the project, 69 animals have been released and more than 127 pups have been born in the wild. Currently, 90% of the free-ranging wolves in eastern North Carolina at Alligator River Wildlife Refuge are wild-born, which illustrates the biological success of the restoration efforts (Phillips 1997; Waddell 1997). Pup mortality has been higher in litters born to captive-born females than to wild-born females, 25% and 6%, respectively (Bergman 1997). To continue genetic exchange, new techniques such as cross-fostering captive-born pups to wild litters have been developed at the Point Defiance Zoo and show promise (W. Waddell pers. comm.). The programme has had its share of problems and controversies, including the question of the genetic purity of the red wolves released into the wild (Wayne *et al.* 1998), but the success of the reintroduction programme has paved the way for additional programmes.

Mexican wolf

The Mexican wolf is the rarest and most genetically distinct subspecies of the grey wolf in North America. Endangered in both the USA (1976) and Mexico (1974), the Mexican wolf was considered extirpated from the USA in 1970. It is considered extremely rare or extinct in Mexico, where it has not been seen in the wild since 1980. The USFWS conducted trapping efforts in Mexico between 1977 and 1980, capturing five individuals, considered to be the last Mexican wolves in the wild (Siminski 2002). Three zoos in the USA volunteered to provide space for the wolves (Lindsey and Siminski 2003) and a captive breeding programme was begun in collaboration with the USFWS Mexican Wolf Recovery Team. Management of the breeding programme became the responsibility of the holding institutions in 1985, with the formation of the Mexican Wolf Captive Management Committee (Lindsey and Siminski 2003). The Captive Management Committee worked in collaboration with the governmental agencies in both countries, the Instituto Nacional de Ecología de la Secretaría del Medio Ambiente Recursos Naturales y Pesca in Mexico and the USFWS in the United States.

In 1993, the Captive Management Committee reorganised as an SSP7, becoming the first bi-national SSP7 programme. The mission of the Mexican Wolf Programme has been to support the re-establishment of the Mexican wolf in the wild through captive breeding programmes, public education, and research (Siminski 2002). Extensive genetic testing resulted in two additional captive populations of Mexican wolves to be certified as 'pure', one population originated from the Chapultepec Zoo in Mexico City and the other from private ownership in the USA (Shields *et al.* 1987; Weber-Rodriguez 1989; Fain *et al.* 1995; Hedrick 1996; Garcia-Moreno *et al.* 1996). These two populations were integrated into the breeding programme, which in 2002 consisted of 241 individuals in 47 institutions (USA and Mexico).

All holding institutions follow captive management techniques that were designed for the goal of reintroduction: decreased human contact and large exhibits with natural conditions to increase the chance of exposure to prey. Immediately prior to release, selected wolves are moved to pre-conditioning pens that are not open to the public, for acclimatisation and further evaluation (Lindsey and Siminski 2003). The first reintroduction of Mexican wolves in the USA occurred in 1998 at the Apache National Forest in Arizona when 11 captive-born wolves were released from their acclimation pens into the wild. Additional releases have occurred on the reservation of the White Mountain Apache Tribe in Arizona on land contiguous with the Apache National Forest. The initial releases were not without problem, and several wolves were shot by hunters. However, as of 2003 there are at least 32 free-ranging Mexican wolves in Arizona and New Mexico. Releases in Mexico are planned for 2004.

Island fox

The primary justification for establishing captive populations of island foxes was protection. Dramatic declines in the numbers of free-ranging foxes on four of the California Channel Islands resulted in recommendations that foxes be captured and maintained in pens on the islands until the cause(s) of decline could be determined and mitigated. Captive breeding was a secondary objective, meant to increase the number of animals until release would be feasible (Coonan 2002). Captive breeding of the wild-caught animals has been generally successful, although some genetically important individuals have not reproduced (Coonan and Rutz 2003).

The foxes in captive facilities on the islands cannot be transferred to mainland zoos due to regulations against them being moved back to the islands later for release after being held on the mainland. The concern centres on the potential for diseases or parasites being introduced to the islands, which could hinder recovery efforts. There are, however, island foxes from San Clemente Island in six mainland zoos. This exception highlights the different histories of the foxes from the various islands. First, the foxes inhabiting each of the six islands are considered separate subspecies, so are being managed separately, and only four of the six subspecies are currently considered endangered by the USFWS. The foxes on San Clemente Island are not among the four considered endangered, but were found preying on the endangered San Clemente loggerhead shrike. To protect the shrikes, foxes were captured, and some were transferred to mainland zoos where they serve primarily an educational function as the focus of a programme to inform Californians about the plight of their cousins on the other islands.

However, unless reintroductions can proceed in the near future, the growing number of foxes in the island

captive facilities may necessitate the transfer of at least some individuals to mainland zoos. A recommendation from the most recent Island Fox Working Group Meeting (Coonan 2002) is to establish goals for the San Clemente foxes in mainland zoos. Possible outcomes might be phasing out San Clemente Island foxes to create space for other subspecies for a breeding programme or to house surplus non-breeding foxes from the islands. The Canid and Hyaenid TAG will participate in these discussions.

16.2.4 Research

Although we may never be able to reintroduce maned wolves, bush dogs (*Speothos venaticus*) or the other canid species back into nature, studies of captive animals can provide insight into their biological processes that can inform conservation programmes. Many techniques, such as hormone assays, and medical treatments, such as vaccine regimes, can be tested and validated in captivity for later transfer to the field. Knowledge of husbandry techniques can also be of direct benefit to wild populations. When the decision was made by the National Park Service Island Fox Recovery Team to remove animals from the wild, information on pen design, shift doors and nest box design developed for fennec and swift foxes were used to design the captive facilities.

The behavioural needs of canids in captivity present a challenge due to limited space and the need to manage the genetics of the small population, which results in the creation of artificial pack situations. Research documenting responses to introductions of related and non-related individuals are being monitored in African wild dogs as a method to improve reintroduction outcomes (K. McCreery pers. comm.). Basic information on mating behaviour and parturition can be recorded in the captive setting with time-lapse infrared video cameras capable of recording 24hrs without the need for human presence. These systems have been used to provide valuable data on parturition success in the Mexican wolf (S. Lindsey pers. comm.) and are being used to study courtship and mating behaviour of the island fox.

Genetic management

Zoo populations are small even when managed collectively, and small populations require intensive effort to manage genetic diversity. The effort by zoos to keep accurate studbooks and manage their captive populations for retention of genetic diversity creates opportunity for studies of theoretical population management (Ryder 2003). Although there may be a great deal of debate regarding this concept, wildlife reserves and sanctuaries are becoming more isolated and the insight gained from zoos in the management of small population genetics can greatly assist the management of free-ranging wildlife in the future.

Reproductive physiology

Captive populations have also been the focus of studies of reproductive physiology and the development of assisted reproductive techniques, such as semen cryopreservation and artificial insemination. Very little is known about basic reproductive parameters for most canids. Grey wolves, coyotes, and red (*Vulpes vulpes*) and Arctic foxes (*Alopex lagopus*) are the exceptions, primarily because reproductive tracts were often examined when animals were killed as part of predator control programmes. In addition, because the two fox species are also bred for fur production, some aspects of their reproductive physiology have been thoroughly studied. Data on basic reproductive parameters, such as the number of oestrous cycles per year, the extent of reproductive suppression of subordinates, and other life history data, have not been systematically documented for most other canid species. Such information is important not only for managing captive and free-ranging populations, but forms the basis of Population and Habitat Viability Analyses (PHVA).

Captivity can provide the ideal setting for reproductive studies, because the animals are likely to be habituated to human presence. Also, samples for hormone monitoring can be more easily obtained than from wild individuals. Hormone patterns during reproductive cycles have been published for fennec fox (Valdespino 2000; Valdespino *et al.* 2002) and red wolf (K. Goodrowe pers. comm.), are ongoing for bush dog (K. DeMatteo pers. comm.), island fox and New Guinea singing dog (*Canis hallstromi* – see section 9.1, p. 223) (C. Asa and J. Bauman pers. comm.) and have begun in maned wolf (N. Songsasen pers. comm.). Semen traits have been characterised and extensive development of semen cryopreservation techniques has been conducted for both red (Goodrowe *et al.* 1998, 2001) and Mexican wolves (Asa 2001; Musson 2001; C. Zindl unpubl.). Semen from genetically valuable individuals is maintained in semen banks for both of these species as part of the captive management plan.

Although records from ISIS and studbooks are not collected as part of prospective research, they do provide a source for some basic life history data, such as age of first reproduction, reproductive life span, inter-birth interval, seasonality and litter size. However, caution must be used when examining these data as captive management practices may affect results. For example, in most cases data on mate access are not recorded so inter-birth interval could be over-estimated.

In addition, if small isolated populations are to be managed genetically, basic assisted reproductive techniques such as artificial insemination could be an important substitute for translocating animals. Unfortunately, manipulation of the canid reproductive cycle has proven more difficult than in most other species. Improvements in the success rate of artificial insemination in the domestic dog are encouraging, but the extensive handling required to

determine the time of oestrus and ovulation plus repeated inseminations make the technique impractical or even unacceptable for application to wild canids. However, recent success with inducing oestrus and ovulation with a short-acting GnRH agonist (Ovuplant7: ZooPharm, Ft. Collins, Colorado) in grey wolves, followed by either natural mating or artificial insemination, resulted in the birth of live pups (C. Asa and K. Bauman unpubl.). This technique provides an alternative to monitoring the hormonal changes associated with ovulation and permits inseminations to be properly timed.

More advanced methods of assisted reproduction, such as embryo transfer and in vitro fertilisation, are also more difficult in canids than in other species and require even more handling and manipulation than does artificial insemination. For assisted reproduction beyond semen cryopreservation or artificial insemination to become part of recovery or management programmes, considerable research and development are necessary. However, such advanced methods may not contribute substantially more than artificial insemination.

Another challenge to reproductive management or monitoring in canids has been detecting pregnancy, because of the obligate pseudopregnancy that follows ovulation in females that do not conceive (Asa 1996). This pseudopregnancy has previously been indistinguishable from pregnancy with assays for progesterone, the steroid hormone characteristic of pregnancy. However, an assay for relaxin, a peptide hormone elevated by mid-gestation in pregnant but not pseudopregnant females, has recently been validated for generic grey and Mexican wolves and for island foxes (J. Bauman pers. comm.). Developed for domestic dogs (WitnessRelaxin, Synbiotics Corp. USA), this assay may also be accurate for pregnancy detection in other canids.

Contraception and population control

Contraception has been used successfully in zoos for more than 25 years for genetic management and to limit production of surplus animals, but attempts to extend the application of contraceptive or sterilisation techniques to free-ranging populations have been problematic. Difficulties include delivery and species specificity, but application to canids presents problems beyond those encountered with other mammalian taxa. The commonly used progestin-based contraceptives are effective but are associated with potentially lethal side effects (reviewed in Asa 1996). The zona pellucida vaccines that have proven successful in some free-ranging ungulates appear to cause irreversible damage to the ovaries of canids (Mahi-Brown *et al.* 1988). However, hopes that a single injection of the vaccine might be effective as a chemosterilant have not been successful (J. Kirkpatrick pers. comm.).

Although administration of contraceptives or sterilants to free-ranging animals presents additional challenges, the results of research and monitoring efforts in captive

populations can provide basic information on efficacy and safety. Currently, a promising new contraceptive alternative for canids is being tested in zoos. The GnRH agonist implant Suprelorin7 (Peptech Animal Health, Australia), which appears to be safe and effective for up to one year in both domestic (Trigg *et al.* 2001) and African wild dogs (Bertschinger *et al.* 2001, 2002), may provide an alternative for free-ranging as well as captive canids. Trials with two other canid species (grey wolves and bush dogs) have been less successful, although failures are believed to be due to inadequate dose (Bertschinger *et al.* 2001). As an alternative to reversible contraceptives, a technique for chemical vasectomy developed in domestic dogs (Pineda *et al.* 1977) can achieve permanent sterilisation in the field without surgery.

Immobilisation

Although small canids can often be manually restrained for quick procedures, large canids must be chemically restrained for safe handling. The dissociative anaesthetics ketamine and Telazol (tiletamine plus zolazepam) are most frequently used with captive canids (for reviews see Kreeger 1999 and Kennedy-Stoskopf 2003). Many immobilisation drugs or drug combinations are often first tested in captivity, especially on species where little field research has occurred, such as bush dogs. Some modification may be needed to decrease the induction time for free-ranging animals, but should be determined on a species-by-species basis.

16.2.5 Animal health and nutrition

Captive canids are routinely vaccinated for disease, and data on reactions are recorded in the medical records. New vaccines or vaccine regimes can be tested in the captive setting as well, since animals can be closely monitored and blood samples taken at intervals to evaluate serological titers. One of the outcomes from the 1996 African Wild Dog Master Plan meeting was the recommendation from regional coordinators, as well as representatives of the Canid Specialist Group, to utilise the captive population for vaccine testing for distemper and rabies. Distemper vaccine testing has also been requested for the island fox, perhaps using the generic gray fox (*Urocyon cinereoargenteus*) as a model.

In addition to data on vaccination and immobilisation protocols, results of blood chemistries are recorded and submitted to ISIS to the physiological reference database. Examples of these reference values for both serum chemistries and haematological parameters can be found in Kennedy-Stoskopf (2003).

Nutrition is an important component of animal health. Recent advances in the formulation of meat-based diets have decreased bacterial load and increased palatability in carnivore diets (Allen *et al.* 1999). Specific nutrition concerns of canids in captivity are rare, but maned wolves have been

problematic, with poor body condition and coat, dental problems and a poor reproductive rate (Bush 1980). Cystinuria, or excessive levels of cystine in the urine, has been documented in maned wolves. Originally cystinuria was thought to be linked to diet in captive maned wolves, but it has also been found in wild maned wolves (M. Rodden pers. comm.). Research on dietary factors and the formulation of new diets are ongoing (Childs *et al.* 2001).

16.2.6 Funding

The financial contribution zoos make to canid conservation in the form of support of captive breeding programmes and direct support for *in situ* projects is substantial. The Red Wolf SSP7 programme was the subject of a study by a Cornell University economist William Rosen, who estimated that zoos spend about \$351,000 per year on facilities, food and staff (Bergman 1997). If that figure is multiplied by the other SSP7 programmes managed at AZA facilities, then that would suggest that AZA zoos spend more than \$1,404,000 per year on canids alone. Additionally, zoos fund numerous field research and educational initiatives each year, although no figure exists for the amount spent directly on canid programmes. AZA zoos have contributed over \$10,000 in support of African wild dog field projects and \$11,000 for the Ethiopian Wolf Population and Habitat Viability Analysis (PHVA) meeting.

16.3 Structure of captive canid programmes

The programmatic structure of all captive conservation programmes is dictated by the regional zoo associations. Three of these, the American Association of Zoos and Aquariums (AZA), the European Association of Zoos and Aquariums (EAZA) and the Australasian Regional Association of Zoological Parks and Aquaria (ARAZPA), have well-organised and active programmes for canids. All have recently published Regional Collection Plans for canids. It should be noted that other programmes do exist in zoos in other places of the world (for example, Brazil, details of which are not published and, therefore, could not be included in this chapter). Since the movement to organise individual zoos into cooperative components of a conservation network began in North America, we will rely heavily on the AZA model to describe how the organisation of captive programmes facilitates the support of canid conservation.

The studbook forms the foundation for all captive programmes. Accurate pedigree data are vital to genetic management of the captive population, and breeding decisions are based on the genetic and demographic information contained in the studbook.

16.3.1 Species Survival Plans

The SSP focuses captive breeding and conservation efforts at the species level. The main function of an SSP is coordination of captive efforts. For example, the AZA Mexican Wolf SSP meets annually with the counterpart Mexican zoo team to discuss progress and problems. Captive wolves in the two countries are managed as one population, with international transfers becoming more common as genetic pairings are recommended between wolves residing on different sides of the border. Representatives from USFWS also attend these meetings, as well as similar ones for the red wolf, to coordinate reintroduction efforts. Although educational programmes can be targeted at the taxon level, the majority are facilitated through the individual SSP programmes. These programmes typically include an explanation of the conservation need and the status of both the wild and captive populations. Educational materials, especially *ex situ* programmes are also vital for schools in range countries. All information necessary to maintain a SSP species in captivity is published in husbandry manuals. These manuals are updated every few years and contain recommendations for housing, nutrition, veterinary care, social groupings, contraception and behavioural needs. Funding for species-specific research and *ex situ* projects is also facilitated through the SSP.

Species level management exists at the highest level for: Iberian wolf (*C. l. signatus*) (EAZA), maned wolf (EAZA, AZA, ARAZPA), bush dog (EAZA), African wild dog (EAZA, AZA), fennec fox (AZA, ARAZPA), red wolf (AZA) and Mexican wolf (AZA). Species-level management also exists for: fennec fox (EAZA), dhole (ARAZPA), African wild dog (ARAZPA) and island fox (AZA). Both EAZA and AZA have dedicated space for the Ethiopian wolf (*Canis simensis*) if the Ethiopian authorities request future support.

16.3.2 Taxon Advisory Group

Zoos have made great strides in efforts to ensure that each species in captivity has a role to play in canid conservation. Decisions regarding which species are kept in captivity are made by the TAG during the creation of the Regional Collection Plan by evaluating all the based on an objective set of criteria.

The AZA Canid and Hyaenid TAG Regional Collection Plan (RCP) was published in 2002. The scope of the RCP includes all canid species, even though some of the species have no history of being in captivity. Taxonomic classifications were adopted from the Canid and Wolf Specialist Groups and excluded subspecies, with the exception of the Mexican wolf, which is of specific conservation concern in North America.

Since zoos have limited space to hold canids, it is vital that the RCP take into account the current number of

spaces available for each species, in addition to the number of projected new spaces. Additionally, captive populations in other regions were assessed, where relevant, to reduce duplication of effort.

Conservation status was taken from three sources: *The 2000 IUCN Red List of Threatened Species* (see <http://www.redlist.org> for the most recent Red List); *Foxes, Wolves, Jackals and Dogs: An Action Plan for the Conservation of Canids* (Ginsberg and Macdonald 1990); and the U.S. Fish and Wildlife Service web site (<http://www.fws.gov>). Two decision trees were constructed, one for species currently held (Figure 16.1) and another for those not currently held in AZA zoos (Figure 16.2). The conservation status of the species in the wild was the most

important selection criterion. Key elements included listing the specific programme contribution, e.g., genetic reservoir for reintroduction purposes, fund-raising or research and whether there was a recommendation for captive programme from field scientists. Priority was given to programmes already established, resulting in lower ranks for populations not in captivity. Discussions after the decision-tree ranking regarding the perceived costs and benefits to each captive programme included existing husbandry skills and interest from member zoos in the species.

Then, based on rank, each species was placed in one of six categories: Species Survival Plan (SSP) with intense, captive management; Population Management Programme

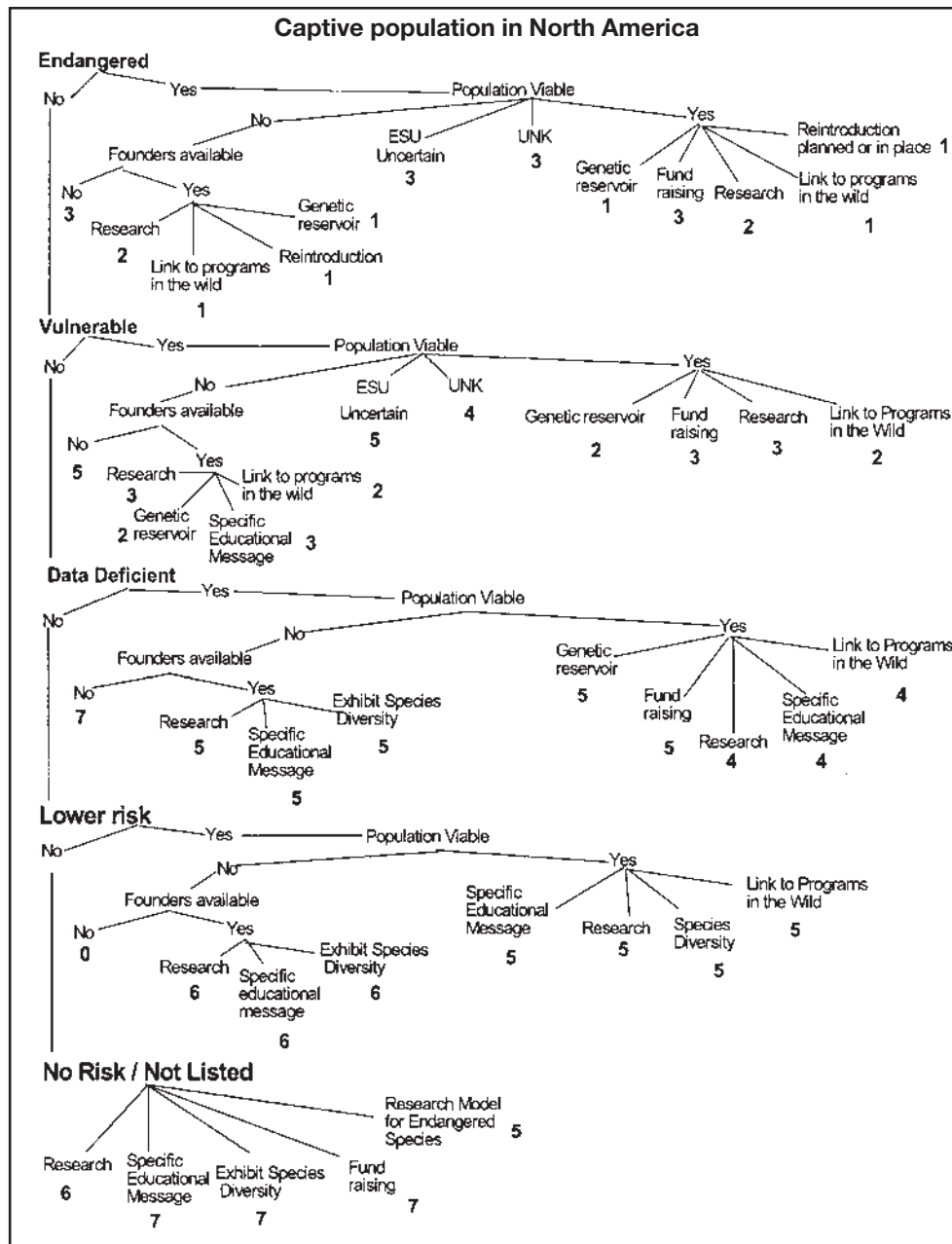


Figure 16.1. Decision tree for canid species currently held in AZA zoos. The decision tree model places selection criteria with the highest priority on the top 'branches'; the lower the 'branch', the lower priority.

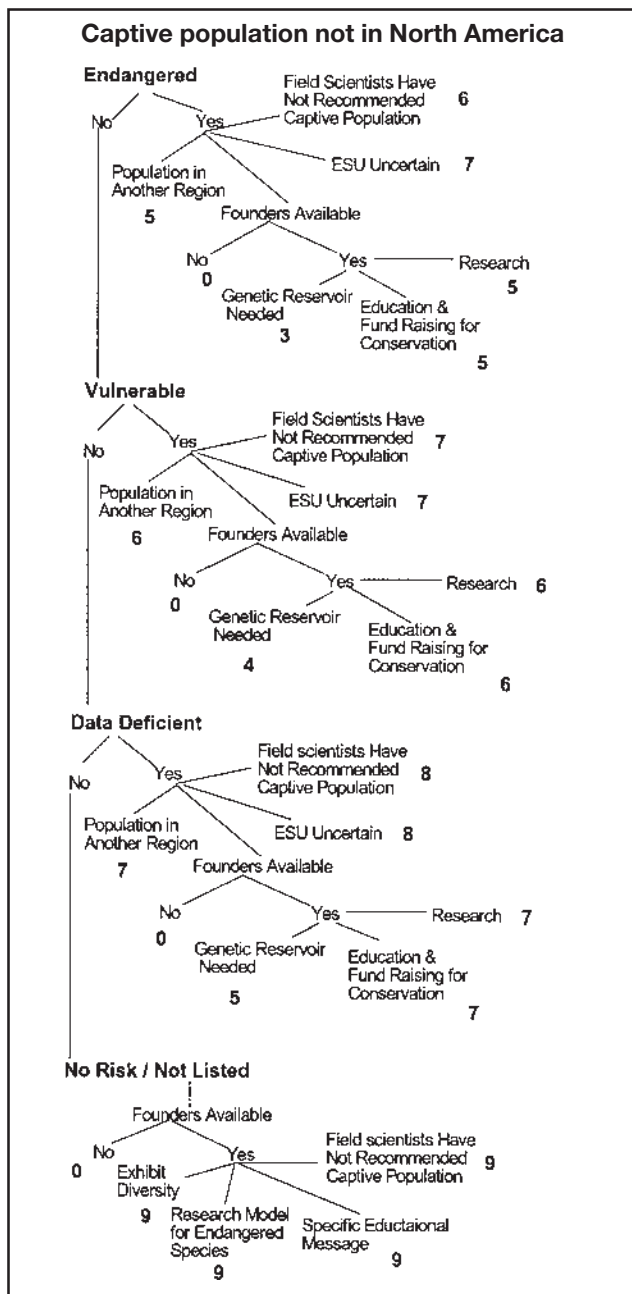


Figure 16.2. Decision tree for those canid species not held in AZA zoos. The decision tree model places selection criteria with the highest priority on the top 'branches'; the lower the 'branch', the lower priority.

(PMP) with long-term captive management, but less intensely managed genetically than an SSP; Display/Education/Research Population (DERP), for species where no genetic or demographic management is needed; Phase-in Population (PIP), for any new species requiring a management programme; Phase-out Population (POP), for species with low conservation status, and whose space could be better utilised to house species of higher conservation status; and No Recommended Programme (NRP), for species with no current management

programme and that, despite threatened status, are not recommended for management. The inclusion of the category for phasing out a species acknowledges that a greater contribution to conservation can be made with fewer, higher quality programmes rather than the more traditional approach of exhibiting the greatest diversity of species within a taxon.

16.4 Conclusions

In his book the *Management of Wild Mammals in Captivity*, Crandall (1964) described the evolution of zoos well by stating that "...the zoological garden is a fluid, moving entity, changing so constantly in concept and execution...". Indeed, zoos have been transformed from places for the public to view the strange and the unusual, to organisations that contribute resources directly to conservation efforts. Today, zoos have realised that captive programmes support conservation in many ways: education, captive breeding, reintroduction, scientific research and funding are all tools that zoos provide.

However, due to limited resources in zoos worldwide and the importance of captive breeding to the conservation of canids, the captive breeding community will need to work even closer with conservation biologists to identify where to prioritise their activities and research. Increased communication between the canid TAG programmes of all zoo associations and the Canid Specialist Group is essential. There has been much effort on the part of zoos to better integrate captive efforts with the needs of conservation in the wild, but work must continue in this area. Continued support of field research initiatives is essential, and not only of species held by zoos in their collections. In addition, there should be increased communication among the TAG programmes of the various zoo associations to further reduce duplication of effort, and to facilitate exchange of ideas and techniques. An important action over the coming years will be to assist zoos in regions of the world with high wild canid density, and/or not covered by a zoo association, through the formation or support of captive programmes.

While it is essential that zoos continue to support basic research in the areas of husbandry, behaviour, genetics, reproductive physiology, contraception and population control, immobilisations, vaccines, animal health and nutrition, and genome resource banks, an important recommendation is the need to create a process in which captive canid programmes can be objectively evaluated. The evaluation should cover the genetic and demographic goals of the captive population, research efforts and contribution and link to field conservation efforts. This will assist zoos in remaining focused on the relationship between the captive population and the conservation needs of wild canids.

Canid Reintroductions and Metapopulation Management

A. Moehrenschrager and M.J. Somers

17.1 Introduction

Human-induced habitat loss, habitat fragmentation, hunting, poisoning, and trapping have led to the extirpation of small canid populations or, in extreme cases, species. However, if sufficient captive or wild animals of a species persist, the potential for the restoration of extirpated populations still remains. Here we examine the successes or failures of canid reintroductions around the world and aim to identify lessons from these programmes that might aid future reintroduction attempts.

In 2002, the IUCN/SSC Reintroduction Specialist Group held a Strategic Planning Workshop where reintroductions were shown to be growing in conservation significance because they:

- are increasing in number;
- attract public attention;
- are regionally important; and
- can use flagship species to facilitate habitat conservation.

Certainly, canids are generally charismatic, ecologically significant, and often sufficiently wide-ranging to be adequate umbrella species for habitat preservation, but the restoration of many species can still be ecologically or politically problematic.

Successful reintroductions require that a number of species-specific, environmental, and bio-political criteria be met (Kleiman and Beck 1994). There should be a need to augment the wild population, sufficient founder stock should be available, and extant wild populations should not be jeopardised by the reintroduction (Kleiman and Beck 1994; Woodford and Rossiter 1994). The species' biology should be well understood, appropriate reintroduction techniques should be known, and sufficient resources should be available for the programme. The original causes for the species' extirpation should be removed and sufficient unsaturated, protected habitat should be available. Reintroductions should conform to legal requirements, be supported by both government and non-government agencies, and have minimal negative impacts on local people (Kleiman and Beck 1994).

Compared to smaller and less wide-ranging species, many canids could be ill-suited for reintroduction because:

- their large home range requirements can only be satisfied

in extensive protected areas which might not be available (Woodroffe and Ginsberg 1998);

- local people frequently oppose the reintroduction of species that prey on domestic livestock or threaten humans (Phillips 1995; Woodroffe and Ginsberg 1999b); and
- the extensive planning and implementation required for reintroductions (Fritts *et al.* 1997) is prohibitively expensive.

In this chapter we aim to delineate aspects of metapopulation theory that have been incorporated into the recovery planning of threatened canids. We also review canid reintroductions to delineate some crucial taxonomic, political, and biological factors that may determine restoration success. Finally, we outline issues that we see as action plan priorities for canid reintroductions in the future.

17.2 Metapopulation management from theory to practice

A metapopulation is a set of spatially isolated groups of individuals that share individuals among them (Wells and Richmond 1995). Numerous canid species are threatened by habitat loss or habitat degradation. One of the main consequences of this is increased fragmentation (Saunders *et al.* 1991) which increases the likelihood of extinctions (Gilpin and Hanski 1991). When habitat is limited, extant and reintroduced canid populations must be managed as a metapopulation to ensure their long-term persistence.

When individuals are moved from one location to another and released to re-establish populations or metapopulations, the scope of such activities may differ depending on programme goals, release techniques, and geographic aspects. We adopt the definition of a **reintroduction** as an attempt to establish a species in an area which was once part of its historical range, but from which it has been extirpated or become extinct (IUCN 1998b). We include wild or captive animals, or a combination of these in our definition. A **translocation** is a deliberate and mediated movement of wild individuals or populations from one part of their range to another with existing conspecifics. **Supplementation** is the addition of individuals to an existing population of conspecifics. **Soft releases** are those releases

where the animals are housed in an enclosure at the place of release for some time before being set free. **Hard releases** are those where the animals are released directly from vehicles or crates without any acclimatisation phase at the place of reintroduction.

Species respond differently to fragmentation and, therefore, an autecological approach has been suggested to the maintenance of metapopulations (Laurance 1991). Differential responses can be due to the landscape pattern, levels of habitat loss (Harrison and Fahrig 1995), life-history, and ecological traits such as diet or vulnerability to predators and competitors (Laurance 1991). The persistence of patchy distributions is related to a number of factors, including rates of local extinction within habitat patches, as well as the frequency of immigration and recolonisation (Lindenmayer and Lacy 1995). It has been shown that seemingly low levels of dispersal can be sufficient to create a stable metapopulation structure (e.g., Simberloff and Cox 1987), which may also allow the viability of canid populations despite relatively small exchange rates between isolated sub-populations.

Four metapopulation parameters have been identified (Hanski 1999) that would characterise canid metapopulation dynamics:

- habitat patches can support locally breeding populations;
- all patches are at risk of extinction;
- recolonisation must be able to occur; and
- the dynamics between patches are asynchronous.

The successful management of sub-populations depends on the minimum viable metapopulation size (Hanski *et al.* 1996). This is the minimum number of interacting local populations necessary for the long-term persistence of a metapopulation in a balance between local extinctions and recolonisations (Hanski *et al.* 1996). Genetic and demographic management of canid sub-populations is essential. Demographic management should aim to control possible negative detrimental factors (e.g., sex ratio variation) as well as declines in population size due to stochastic demographic processes (Foose and Ballou 1988). Genetic management aims to reduce the effects of inbreeding and genetic drift to allow for genetic population viability over time (Lande 1988; Lacy 1997). While metapopulations are governed by extinction and recolonisation rates, reintroduction practitioners have the luxury of artificially choosing immigration sites and numbers through sound metapopulation planning.

17.3 Metapopulation planning

17.3.1 Ethiopian wolf

Ethiopian wolf (*Canis simensis*) surveys by Marino (2003) confirm that the population consists of around 500 subadult

and adult individuals, of which half are found in the Bale Mountains. The survey has also revealed the presence of several other Ethiopian wolf populations in Afroalpine habitat north of the Rift Valley. Newly found populations in Wollo are small in size (between 25 and 50 wolves) and almost all small habitat patches in the Afroalpine units are occupied. Recently the species was extirpated from a 20km² habitat patch in Gosh Meda and with the exception of Bale, and possibly Arsi, all Ethiopian wolf populations may be vulnerable to extinction due to their small population and habitat patch sizes. However, their continued persistence in small populations suggests the species may be relatively resilient (Gottelli *et al.* 2004), although there are no recent examples of recolonisation and recovery after local extinctions (Sillero-Zubiri *et al.* 2000). A metapopulation management approach has been proposed for Ethiopian wolves, although reintroductions are not recommended at this time (Sillero-Zubiri and Macdonald 1997).

17.3.2 Grey wolf

After grey wolf (*Canis lupus*) populations began to recover in southern Canada and after the U.S. Endangered Species Act protected wolves as of 1974, wolves began to recolonise northern Montana naturally. The first case of wolves reproducing in this area was documented in 1986, and by 1993 the population had grown to 88 wolves in seven packs (Fritts *et al.* 1995).

The U.S. Fish and Wildlife Service appointed a Northern Rocky Mountain Wolf Recovery Team to further recover this wolf population (Fritts *et al.* 1997). The subsequent recovery plan identified north-western Montana, central Idaho, and the greater Yellowstone area, totalling about 69,000km², as potential release areas based on prey numbers, sufficient publicly owned land, and a low potential of conflicts between wolves and humans (USFWS 1987). Assessments suggested that the restoration of 10 breeding pairs per release area would create a viable metapopulation (USFWS 1994; Fritts and Carbyn 1995; Fritts *et al.* 1997), if subsequent dispersal connected the release sites and the Canadian wolf population (Boyd *et al.* 1995). Haight *et al.* (1998) modelled wolf subpopulations and concluded that wolves can survive in sub-populations if there is genetic connectivity between them, human persecution is not excessive, and prey is abundant. They used 16 sub-populations and found that as few as two immigrants a year helped maintain high (>80%) site occupancy. The recovery plan suggests that, if recovery-level populations totalling about 300 wolves were reached and maintained for three consecutive years, the species would be removed from ESA protection to be managed by state and tribal wildlife agencies.

Two other grey wolf recovery programmes, guided by their own recovery plans, exist outside the northern



William Campbell

A nine month old male grey wolf watches biologists after being captured and fitted with a radio collar in Yellowstone National Park. This park is the site of the successful Yellowstone Wolf Restoration Project, made possible by conservation policy and government, as well as private, funding. For example, the United States National Park Service called for restoring the human extirpated wolf population under its native species policy. The U.S. Congress provided funds to the Fish & Wildlife Service, National Park Service and U.S. Forest Service, to prepare a statement on restoring wolves to Yellowstone and central Idaho. The Yellowstone Wolf Restoration Project was started in the park, and has become a model in successful reintroduction and canid conservation. Yellowstone National Park, Wyoming, USA, 2003.

Rockies; one in the upper Midwest (USFWS 1992) and the other in the south-western states (Parsons and Nicholopoulos 1995).

17.3.3 Red wolf

Red wolves (*Canis rufus*) were extirpated from almost all of the eastern United States at the beginning of the 1900s because of persecution and habitat loss. Remaining red wolves were trapped in Texas and Louisiana from 1973 to 1980 to establish a captive-breeding programme, which subsequently led to reintroduction attempts in two mainland and three island populations since 1987 (Phillips 1995).

A metapopulation has been proposed for the red wolf. A population viability analysis indicated that to meet metapopulation management objectives, the USFWS would have to establish a population of 550 red wolves. These 550 would be distributed between a captive population of 330 and a free-ranging population of 220 at three or more sites (Phillips 1995; Reading and Clark 1996).

17.3.4 African wild dog

African wild dogs (*Lycaon pictus*) disappeared from 25 of the 39 former range countries in the last 30 years. Only six populations, found in southern and eastern Africa, had more than 100 animals by 1997 (Woodroffe *et al.* 1997). Many reintroductions have been attempted, most of them unsuccessful (Woodroffe and Ginsberg 1999b). Following recommendations in the IUCN Wild Dog Action Plan (Woodroffe *et al.* 1997) a metapopulation approach to African wild dog conservation in South Africa was proposed by Mills *et al.* (1998). The primary objective was to establish a second South African, artificially managed metapopulation consisting of eight packs in small conservation areas by 2007. Potential reintroduction sites were identified and an adaptive management model was created to implement metapopulation recommendations.

The South African Wild Dog Action Group (WAG-SA) (renamed as the South African Wild Dog Advisory Group in 2003) has been meeting 3–4 times per year since 1997 to oversee the formation of new subpopulations such as those in Venetia-Limpopo, Pilanesberg and Karongwe. Mills (2003) argued that wild dog restoration is only feasible in sufficiently large reserve areas and only if biodiversity conservation is a primary objective for these sites. He suggests that potential metapopulation reserves should be ranked according to their willingness and/or ability to allow ecological processes to occur that would benefit African wild dogs. In priority reserves, metapopulation management should simulate natural conditions as closely as possible. Since wild dog numbers naturally fluctuate (Maddock 1999), subpopulation viability assessments must also incorporate reproductive rates, genetic status, and the potential effects of supplementing single-sex juvenile groups (Mills 2003).

17.4 Review of canid reintroduction programmes to date

Canid species comprised 13% of carnivore reintroductions and translocations reported by 2000, compared to mustelids (58%), felids (22%), ursids (4%) and hyaenids (3%) (Breitenmoser *et al.* 2001). Here we review the current status of canid translocation and reintroduction attempts.

17.4.1 Grey wolf

In at least three attempts grey wolves have been released for purposes other than reintroduction. Experimental releases of four captive wolves in Alaska (Henshaw *et al.* 1979) and five translocated wolves in Michigan (Weise *et al.* 1979) were unsuccessful. In Minnesota, 107 grey wolves were captured at or near farms and successfully

translocated into forested areas where other wolves may have been resident already (Fritts *et al.* 1984).

In Georgia (western Asia) 22 wolves were pre-conditioned to avoid humans and to avoid livestock before a reintroduction attempt. Over four years of monitoring, survival rates were high and two generations of descendents were produced (Badridze 1999).

Grey wolf reintroduction to Idaho has been successful after 15 translocated wolves were released in 1995, 20 in 1996, and one in 1998. Extirpated from the state before 1995, ≥ 122 wolves including 10 reproductive packs were estimated to exist in the wild by 1998 (Bangs *et al.* 1998). The reintroduction of wolves to Yellowstone National Park has also been successful. From 1995 to 1997, 44 wolves were soft-released and by 1998 the Greater Yellowstone population consisted of 116 wolves with seven packs that produced 10 litters of pups (Bangs *et al.* 1998). Within four years of the reintroduction programme, 133 pups had been born in 29 litters in the greater Yellowstone area. In the summer of 2002, at least 216 free-ranging wolves could be found in this region with about 14 packs (132 individuals) holding territories primarily within the park and 14 packs (84 individuals) primarily outside its boundaries (Smith *et al.* 2003). Despite the short interval since reintroductions began, the success of the Yellowstone programme is already initiating debates about the potential de-listing of Yellowstone-area wolves under the U.S. Endangered Species Act (Pyare and Berger 2003).

17.4.2 Red wolf

Analyses of 320 red wolf releases indicated that release success was primarily dependent on the degree of wolf habituation in captivity and landscape manipulation by humans (Van Manen *et al.* 2000). Eight years of reintroduction releases in Great Smoky Mountains National Park of North Carolina and Tennessee were unsuccessful, as juvenile survival was low and wolves failed to establish home ranges within the protected area.

Red wolf restoration to the Alligator River National Wildlife Refuge in north-eastern North Carolina has been successful to date with an estimated population size of 100 individuals of which 63 were still radio-collared in December 2002. At that point, approximately 160 red wolves also existed in 37 captive facilities. Recently, reintroduction methods have been further refined as captive pups have now been successfully fostered by wild parents at the release site (American Zoo and Aquarium Association 2003). Currently, red wolves range across an area of approximately 6,000km² in North Carolina including private and public lands (Phillips *et al.* 2003). One of the greatest remaining threats to red wolves is hybridisation with coyotes, but plans are being implemented to reduce interbreeding and remove hybrids (Kelly 1999).

17.4.3 Mexican wolf

The probable historic range of the Mexican wolf (*Canis lupus baileyi*) included portions of central and northern Mexico, western Texas, southern New Mexico, and south-eastern and central Arizona. The Mexican wolf was extirpated from the wild in the United States by the mid-1900s (Parsons 1998), fewer than 50 adult breeding pairs were estimated to exist in Mexico in 1978 (McBride 1980), and it appears unlikely that viable populations, if any wolves at all, currently remain in the Mexican wilderness (Parsons 1998). The primary factors responsible for this decline were extermination campaigns using traps, digging pups from dens, shooting, and poisoning, which were often encouraged by public or private bounties.

Five Mexican wolves captured in Durango, Mexico, between 1977 and 1980 were used to found a captive-breeding population. Subsequent breeding efforts (reviewed in Chapter 16) led to a captive population of over 200 individuals in 40 zoos and wildlife sanctuaries throughout the USA and Mexico by 1999 (Parsons 1999). The primary goal of the reintroduction effort is to restore a self-sustaining population of about 100 Mexican wolves distributed across 12,950km² in the Blue Range Wolf Recovery Area of south-eastern Arizona and south-western New Mexico (Paquet *et al.* 2001).

In March 1998, 11 wolves representing three family groups were soft-released into Arizona's Apache National Forest on 29 March 1998 (Parsons 1998). Two adults and three subadults were shot and three wolves were captured and returned to captivity. By 1999, the free-ranging population consisted of at least 24 Mexican wolves in five packs, and one wild-born litter whose size was not known (Parsons 1999). Subsequently 45 additional individuals were released by March 2001 (Paquet *et al.* 2001). In May 2001, 28 individuals were reported to exist in the wild and five litters had been produced during that spring. Paquet *et al.* (2001) recommended that releases should continue with major alterations to the goals, procedures, and organisational aspects of the programme.

17.4.4 African wild dog

The first successful wild dog reintroduction was into Hluhluwe-Umfolozi Park (HUP) from 1980–1981 (Maddock 1999). A mixture of 24 wild-caught and hand-reared wild dogs were released, which persisted with a mean annual density of 19.9 wild dogs in the 960km² reserve. In 1986, four semi-tame wild dogs were released into HUP, which left the park (Maddock 1999). As numbers declined after 1993, another three wild-caught males and one female were introduced in 1997. The pack produced 12 pups in 1998 but it split up in 1999 after the alpha female died. The two adult males left the park and roamed widely in other reserves and game ranching areas before one was

found dead and the other disappeared (Somers 2001). Of the 1997 reintroduction, only one male (brought in as a yearling of the alpha female) still survived in 2003. In 2000, two females were introduced which bonded with two existing males from the original pack. As of March 2003 there were two packs, both of which have had pups in 2001 and 2002. A third pack, consisting of two males and two females from the Limpopo Province, is currently awaiting release with an unrelated adult female and three juvenile males. This introduction will be the first where unrelated animals of the same sex have been combined and released.

There were three attempts to reintroduce wild dogs into Etosha National Park, Namibia, from 1978 to 1990 (Scheepers and Venzke 1995). In 1978, six hand-reared yearlings were released which died within four months, mainly due to starvation (Scheepers and Venzke 1995). In 1989, five adult captive-bred wild dogs were released, but all died of unknown causes within three months. After the success of the reintroduction of wild dogs into Hluhluwe-Umfolozi Park, it was decided to try again in 1990. Five captive-bred males (including three adults) and eight females (three adults) were to be released. One adult male and female escaped while being transported to Etosha. The female was shot a week later on a farm. The male formed a pack with two domestic dogs which hunted together until the wild dog was trapped in a reserve 350km away. He was returned to the released wild dog pack, which subsequently killed him with bites on the spine and neck. The remaining animals struggled to hunt and lost body condition. Four died of rabies, six were killed by lions and one disappeared.

Six wild dogs were reintroduced into Tsavo West National Park, Kenya, in 1997. They disappeared after eight days, and eventually all animals were killed (Kock *et al.* 1999). In 1986, nine captive-raised wild dogs were released into the Matetsi Safari Area in Zimbabwe and then shot by a local farmer. In Zimbabwe wild dogs have been captured and released successfully in safer areas (G. Rasmussen pers. comm.).

In 1975, three males and two females were reintroduced into South Africa's Kalahari National Park which soon split up and disappeared (Frame and Fanshawe 1990; cited in Woodroffe and Ginsberg 1999b). In 1992, seven male and seven female wild dogs were reintroduced via soft release to Venetia-Limpopo Nature Reserve, South Africa. They bred that year but later left the reserve and some were found poisoned on farmland (Van Heerden 1992; Woodroffe and Ginsberg 1999b). In January 2002 nine wild dogs were reintroduced, which subsequently bred.

In 1995, three wild-caught and three captive-bred individuals were reintroduced via soft release into Madikwe Game Reserve, South Africa. They produced two litters of pups, however, 18 of 21 wild dogs died of rabies in 1997. In January 1998 three captive-bred females and two wild-

caught juvenile males were reintroduced, and in July 1998 two captive-bred and two wild-caught males were added to the wild population. In 2000 new males were added and have since disappeared. In February 2000 another rabies outbreak killed eight of 11 unvaccinated pups but none of the vaccinated adults. There are now three successfully breeding packs in Madikwe, and some individuals have been removed to supplement other release areas, including Hluhluwe-Umfolozi Park.

Nine wild dogs were reintroduced by soft release into Pilanesberg National Park, South Africa, in 1999 (van Dyk and Slotow 2003). This release again demonstrated that a combination of wild-caught and captive-bred wild dogs could successfully be used for wild dog reintroductions. In 2002 another two females were bonded with two present males and two packs now exist in the park. These packs have bred on numerous occasions.

In 2001, one wild-caught male and two captive-bred females were reintroduced by soft release into Karongwe Nature Reserve, South Africa. The three successfully bred in 2002.

Captive wild dogs are presently awaiting reintroduction into Marakele National Park and reintroductions are being planned for other areas such as Addo Elephant National Park and Greater St Lucia Wetland Park, South Africa.

17.4.5 Swift fox

The swift fox (*Vulpes velox*) is native to shortgrass and mixed-grass prairies of the Great Plains in North America (Egoscue 1979). Historically, the swift fox was considered an abundant predator of the prairies, but numbers were severely depleted by the late 1880s and early 1900s. On the northern limit of its range, swift foxes were present in the Canadian provinces of Alberta, Saskatchewan, and Manitoba. The southern species' boundary was New Mexico and Texas in the United States. Historical records also exist for areas in Montana, Wyoming, North Dakota, South Dakota, Nebraska, Kansas, Colorado, and Oklahoma. Current estimates for the USA suggest that swift foxes are located in 39–42% of their historic range depending on conservative vs. liberal estimates of historic range and the time span of records that are considered (Sovada and Scheick 1999). In Canada, the last recorded specimen was collected in 1928 and a single sighting was made in 1938 (Soper 1964), after which the species was considered nationally extirpated.

As part of a national reintroduction programme, 942 swift foxes were released in Canada from 1983 until 1997 (see section 4.6). Translocated foxes that were monitored from 1994–1998 had higher survival rates than previously monitored captive-bred foxes, and similar survival rates to resident, wild-born foxes (Moehrenschrager and Macdonald 2003). In 1997, the Canadian population was

estimated to have approximately 192 and 89 in respective subpopulations. Of foxes that were live-captured from 1994–1998, 88% were born in the wild within the reintroduced population (Moehrenschrager 2000; Moehrenschrager *et al.* 2003). By 2001, the number of individuals trapped on replicated townships had increased significantly, and the known distribution of swift foxes increased three-fold since the previous census. While the population was previously fragmented in Canada and sparse in Montana, the population is now connected because gaps within the known distribution are smaller than maximum dispersal distances of this species (Moehrenschrager and Moehrenschrager 2001). This increased connectivity might decrease the likelihood of extinction through coyote predation or red fox competition, which are the primary threats (Tannerfeld *et al.* 2003; Moehrenschrager *et al.* 2004), but increases the possibility of disease transmission throughout the population.

17.5 Lessons learned

17.5.1 Socio-political factors can make or break canid reintroduction programmes

Several authors have pointed out that valuational and organisational aspects are at least as critical for carnivores as biological parameters (Beck *et al.* 1994; Reading and Miller 1995; Miller *et al.* 1996; Reading and Clark 1996; Breitenmoser *et al.* 2001). Since many canids require large home ranges in protected habitats and many prey on livestock or commercially hunted species, stakeholders such as landowners, hunters, the resource-extraction industry, indigenous communities, regional and federal governments, and conservation organisations may have special interests surrounding the protection of canids. Recovery planning that is inclusive, interdisciplinary, and effective is difficult to achieve, but critical to the protection of canids (Boitani *et al.* 2004).

One of the most crucial aspects determining the success of canid reintroductions is the support of affected landowners, or mitigation measures to placate those that resist such efforts. Support for a possible reintroduction of wolves to New Brunswick, Canada, was lowest for sampled individuals that were hunters, feared wolves, or had low levels of formal education. The primary reason for opposing reintroductions was that deer availability for hunting would decline (Lohr *et al.* 1996). Opponents to a possible reintroduction of wolves into Colorado, USA, expressed concern about wolf attacks on livestock, financial losses to ranchers, wolves threatening residential areas, and large losses of deer or elk. Proponents believed wolf reintroduction would control deer, elk and rodent populations, restore the environment, and help educate the public about wilderness (Pate *et al.* 1996).

Strong opposition from some factions stalled the reintroduction of wolves to Yellowstone National Park and central Idaho for two decades, until a proposal to reintroduce wolves was accepted as long as it was deemed ‘non-essential-experimental’ under the United States Endangered Species Act (Fritts *et al.* 1997). Nevertheless, a group comprised mostly of farmers and ranchers filed a lawsuit to stop this reintroduction. Similarly, the New Mexico Cattle Growers Association filed a lawsuit in the U.S. District Court of New Mexico to stop the reintroduction of Mexican wolves (Parsons 1999).

Careful management of released red wolves and the emerging population involved close public consultation, which has led many landowners to allow wolves on their properties (Phillips *et al.* 1995). Defenders of Wildlife, a non-governmental organisation, has compensated ranchers for livestock losses caused by reintroduced Mexican wolves and grey wolves in Yellowstone. In assessing the future of the Mexican wolf, Paquet *et al.* (2001) concluded that human attitude is the primary factor that will determine the viability of this species.

17.5.2 The taxonomy of historical and potential source populations may determine the feasibility and magnitude of reintroduction programmes

As we attempt to discern which species or subspecies require conservation action, taxonomic classification is of paramount importance. Imperilled species are more likely to receive funding, research, and political protection than subspecies, which in turn are more likely to receive protective measures than geographically-depleted populations of generally abundant species. The likelihood that reintroduction or translocation programmes will be implemented depends directly on the resolution of genetic questions. This, however, is at present usually of more concern in developed countries where detailed genetic data are available for most threatened species.

At the beginning of the swift fox reintroduction programme in Canada, critics cautioned that animals from the central USA should not be used to re-establish northern populations because of a possible mixing of subspecies (Stromberg and Boyce 1986). Reintroductions continued because others doubted such subspecies existed (Herrero *et al.* 1986), and later testing illustrated that these original subspecies designations are likely unwarranted (Mercure *et al.* 1993). The designation of San Joaquin kit foxes as a distinct subspecies (*Vulpes macrotis mutica*) led to federal listing under the United States Endangered Species Act. This increased protective measures and resulted in a regional reintroduction attempt, which was unsuccessful as 97% of released animals died (Scrivner *et al.* 1993).

Reintroduction and conservation efforts to protect Mexican wolves are taxonomically supported by the fact

that Mexican wolves are a genetically distinct subspecies (Garcia-Moreno *et al.* 1996). Mexican wolves were found to have a unique Bgl restriction-site polymorphism and, contrary to distance tree analyses, mitochondrial DNA analysis suggests that Mexican wolves are more similar to Old World wolves than North American conspecifics (Wayne *et al.* 1992). By comparison, the classification of red wolves has been problematic. Mitochondrial DNA analyses demonstrated that the red wolf is either a hybrid form, or that it is a distinct taxon that hybridised with grey wolves or coyotes over much of its geographical range (Wayne and Jenks 1991). Further mtDNA and nuclear DNA analyses support the former hypothesis that red wolves originated through coyote-grey wolf hybridisation (Roy *et al.* 1996), likely in the last 2,500 years (Reich *et al.* 1999). Although some proponents still use morphometric analyses to argue that red wolves are not wolf-coyote hybrids (Nowak 2002), the genetic evidence has presented a conundrum for reintroduction efforts. Recent genetic evidence suggesting that wolves in south-eastern Ontario and southern Quebec, Canada, be classified as a new species, *Canis lycaon* (Wilson *et al.* 2000), may be the beginning of similar conservation challenges for these populations.

Through taxonomic reclassification a seemingly abundant population can suddenly be found to be a Critically Endangered species that requires captive breeding, translocation, or reintroduction. The fact that island foxes (*Urocyon littoralis*), previously thought to be gray foxes, are in fact a distinct species composed of six distinct subspecies (Wayne *et al.* 1991b; Wilson and Reeder 1993), has had immediate conservation consequences which have resulted in captive-breeding and translocation programmes. Nevertheless, the island fox is not yet recognised as an imperilled species under the United States Endangered Species Act. Similarly, Darwin's fox (*Pseudalopex fulvipes*) was thought to be a subspecies of chilla (*P. griseus*), but subsequent genetic evaluation has shown that they are a distinct species, and recently listed as Critically Endangered. Given disease and demographic threats to the population, it is now likely that captive breeding will be initiated.

17.5.3 Soft-releases and translocations are effective reintroduction techniques

Captive-bred swift foxes that were released in autumn had higher survival rates than those released in spring (Brechtel *et al.* 1993; Carbyn *et al.* 1994), but translocated swift foxes were more successful than both captive-release treatments (Carbyn *et al.* 1994; Ginsberg 1994). Survival and reproductive rates were compared between 56 swift foxes that had been born in the Canadian release area to those of 29 Wyoming swift foxes that were translocated from Wyoming between 1994 and 1996 and tracked for up

to 850 days after release (Moehrenschrager and Macdonald 2003). Translocated juveniles dispersed less far but survived and reproduced as well as translocated adults, suggesting that juveniles can be used to establish translocated foxes in small, protected areas, while minimising demographic effects on source populations. The fact that survival rates and litter sizes of translocated foxes were similar to those of resident animals indicates that translocation can be an effective reintroduction tool for this species, and possibly other foxes (Moehrenschrager and Macdonald 2003).

Soft releases were used from 1983–1987 and hard releases from 1987 onwards (Herrero *et al.* 1991) because they were less expensive. Radio-telemetry revealed that survival and reproductive success were highest for swift foxes with small dispersal distances, suggesting that measures should be taken to acclimate animals to release sites through soft releases (Moehrenschrager and Macdonald 2003). In Idaho, hard-released grey wolves also ranged widely and did not reproduce in the first year, while soft-released wolves in Yellowstone National Park remained close to the release site and bred successfully (Smith 1999). Survival rates were highest for red wolves that had been wild-reared with a short acclimation period at the release site (Van Manen 1999).

17.5.4 Released canids can adapt quickly to local conditions

Reintroduced canids only establish populations if individuals can establish territories, hunt, avoid predators, find mates, and reproduce. Depending on body size, guild structure, and predator occurrence, the challenges canids face after release differ between species. Pack hunters such as African wild dogs need to develop social and cooperative hunting skills to kill large prey and resist competitive carnivores, while solitary hunters do not. However, comparatively small solitary foragers, such as island, swift, kit, and Arctic foxes that are preyed upon by predators such as golden eagles (*Aquila chrysaetos*) may need to develop predator avoidance strategies.

Red wolf releases that consisted of adults accompanied by young pups tended to form cohesive groups, exhibit fewer wide-ranging movements, and were less likely to experience vehicle-related mortalities (Van Mannen 1999). Released grey wolves can adapt their hunting skills to local conditions. Only two of 41 studied wolves that had been translocated from Alberta and British Columbia, Canada, to Yellowstone National Park had been previously exposed to bison. Eight one-year-old wolves killed an emaciated bison calf 21 days after release, the first adult bison was killed after 25 months during a 9.5-hour-long hunt, and subsequently 14 bison kills were documented over a period of four years (Smith *et al.* 2000). Captive-bred Mexican wolves had no previous hunting experiences in the wild when the first three family groups were released in

1998. Nevertheless, three weeks after their release, three subadult members of one family killed a mature cow elk and evidence was found to suggest that the other two families had also killed adult elk or elk calves (Parsons 1998).

17.5.5 Disease can hamper reintroduction attempts

In the planning of any translocations or reintroductions, the risk of accidental transmission of disease into unaffected populations via released animals must be carefully assessed. Woodford and Rossiter (1994) recommend that veterinary involvement in reintroduction projects should begin as early as possible. They point out instances of inadequate disease risk assessment resulting in expensive failures, and the introduction of destructive pathogens into resident wildlife populations by captive-bred and wild-caught animals. Infectious diseases may have short-term or long-term effects on population size and viability by affecting rates and patterns of mortality or reproduction. Assessment procedures should address infectious agents that released animals may be exposed to, or that they might carry to conspecifics and other susceptible species at the release site (Nowell and Jackson 1996). Although the risk of disease may be deemed higher where domestic dogs are sympatric, in Madikwe Game Reserve wild dogs were killed by a strain of rabies from black-backed jackals (*Canis mesomelas*) (M. Hofmeyr pers. comm.).

17.5.6 Canid restoration can have profound ecosystem effects

Carnivores are thought to be crucial to the maintenance of healthy ecosystems, but the scale of their function is often difficult to evaluate within extant populations. Reintroductions provide unique opportunities to test the impacts of different types of carnivores. While ecosystem impacts of swift fox reintroduction have not been thoroughly explored, the reintroduction of Yellowstone Park wolves elucidates that the re-establishment of a top carnivore can be felt on all ecosystem levels.

The number of coyotes in Yellowstone Park's Northern Range dropped from 80 individuals in 12 packs before wolf reintroduction, to 36 coyotes in nine packs. Within three years of wolf reintroduction, 25–33% of annual coyote mortality was due to wolves, mean coyote pack size dropped from 6 to 3.6 adults, and coyote population size dropped by 55% (Crabtree 1998). Functionally, surviving coyotes have increased vigilance behaviours and altered foraging patterns since wolf reintroductions began (Switalski 2003). While male behaviour was not affected, the vigilance of female bison and elk increased significantly. Among elk this was true for both females with calves, which increased vigilance rates from 20% to 43%, and females that did not have

calves, which increased vigilance from 11.5% to 30.5% (Laundré *et al.* 2001). Changes in elk foraging patterns can even be detected on a plant community level. Elk pellet counts were significantly lower in habitats that wolves used frequently than in rarely used areas. Consequently, aspen sucker height was significantly higher in areas of high wolf-use than regions that wolves used rarely.

17.6 Conclusions

Despite extinction in the wild, taxonomic controversies, ongoing hybridisation with coyotes, and potential conflicts with landowners, the red wolf reintroduction programme is showing ongoing signs of success. Similarly, the grey wolf reintroduction to Yellowstone has been an ecological triumph despite numerous political and legal battles. Reintroduced swift foxes have been de-listed from 'extirpated' to 'endangered' in Canada. Reintroduction attempts of Mexican wolves and African wild dogs are showing increasing promise and the integration of sound metapopulation planning will facilitate population viability in the future. The diversity of biological and organisational challenges that have been successfully overcome to restore canids should truly be celebrated. That said, numerous challenges still lie ahead.

Ginsberg (1994) determined that although breeding success in canids is highly variable and captive populations of many imperilled species are too small, some can be successfully reintroduced if adequate animals, habitat, and funds are available. Since that time only the island fox, for which releases may begin soon, has been added into reintroduction programmes (Timm *et al.* 2002). Although only nine of the world's 35 canid species exist in the USA or Canada, five of the six species involved in reintroductions are from these countries. This raises the question whether canid reintroductions are primarily aimed at species in wealthy countries, while species that are imperilled in poorer regions receive less attention. Reintroductions can be used as a powerful conservation tool to restore canids (Boitani *et al.* 2004), but it is unclear which canids require such assistance in the future. Hence, we propose that the status, restoration need, and feasibility of reintroductions should be addressed for all canid species in the next five years.

Captive-bred animals and, to a lesser extent, translocated individuals, need to develop hunting- and predator-avoidance skills that are pertinent to their release locations. Successful mixed releases of captive-bred and translocated African wild dogs and the recent success of red wolf fostering in the wild, suggest that such techniques can improve the effectiveness of captive-bred animals. While soft-releases have been successfully combined with pre-conditioning experiments in other carnivores (Reading and Clark 1996), canid restoration programmes often

have not. Experimental designs that rigorously test success differences between fostering practices, pre-release conditioning, and release techniques should become integral programmes of canid reintroduction programmes in the future. Moreover translocations of African wild dogs, grey wolves, and swift foxes have been used with great success, but none have thoroughly evaluated the impact of these activities on source populations. If future translocations are used to restore canids, impact evaluations should not only be required in the release areas but also in the source populations.

In the planning phase, or as reintroductions show signs of success, the question continually arises as to how many individuals need to be restored for the programme to be deemed successful (Pyare and Berger 2003). While the answer hinges largely on demographic parameters that determine effective population size, minimum viable

population size estimates are also dependent on genetic parameters. Some argue that a population of 50 individuals is sufficient to avoid short-term deleterious effects of inbreeding depression (Franklin 1980; Soulé 1980), others believe that 500 is sufficient to maintain genetic variability in quantitative characters (Reed and Bryant 2000), while some believe that 1,000–5,000 individuals may be a safer number to strive for (Lynch and Lande 1998). Tremendous resources are required for canid restorations, many stakeholders are affected, and the viability of reintroduced canids needs to be adequately secured over time. For ecological and political purposes, future minimum viable population size targets need to be more specifically defined for canids in general or, ideally, for specific canid taxa. At the least, future canid conservation efforts would benefit from refined estimates of minimum viable population sizes in pack-living versus solitary canids.

Conservation Education and its Relevance to Wild Canids

D. Taylor

“It is not education, but education of a certain kind that will save us.” (Orr 1994)

18.1 Introduction

Conservation education is an extremely complex and varied subject. The issues involved often raise more questions than answers, and the task of developing and implementing education programmes is a daunting one, especially when you consider that some programmes may take decades to achieve their goals. Education, however, is now becoming more widely recognised as one of a number of effective conservation strategies.

One of the key issues in wild canid conservation is conflict mitigation (see Chapter 13), and trying to implement strategies to foster co-existence. Education has a key role in conflict mitigation, but what is less clear is how to develop, implement and evaluate effective conservation education strategies. There is a dearth of practical information available on this subject, and a lack of multidisciplinary cooperation between all the stakeholders involved.

A further problem is that education programmes are often developed and implemented by people who have no formal training in education as a discipline. Biologists working with individual species may have expertise in their chosen field, but are not usually specialised in both the disciplines of science and education. Non-governmental organisations (NGOs) rely heavily on volunteers who also may not have any formal academic qualifications as educators.

When developing conservation strategies, a number of moral and ethical issues also need to be considered. Not least of these is the reason(s) for conserving a certain species and/or habitat, which species should be given priority, and who should make the decisions – conservationists, politicians, the international community or the indigenous people.

Some of these issues might at first seem beyond the scope of a chapter on conservation education, but the key issue here is: *Who needs educating, and why?*

- Is it the hunters and farmers with the aim of fostering co-existence in human/carnivore conflict situations?
- Is it the policy-makers in order to lobby for legislation to protect wild canids?
- Is it the general public in order to bring about attitude and behaviour changes at a societal level?

- What about education and training programmes for conservationists – again with the aim of fostering co-existence?

The purpose of this chapter is to highlight some of the issues that need to be considered when developing a wild canid conservation education programme, and to provide some general practical guidelines.

18.2 Defining education

As conservationists, we need to be clear on what we mean when we use the term *education*? It is a term that is frequently cited in conservation literature and at biology conferences and symposia, and often without any clear definitions being offered. For instance, is a communication or awareness-raising initiative really an ‘education’ programme?

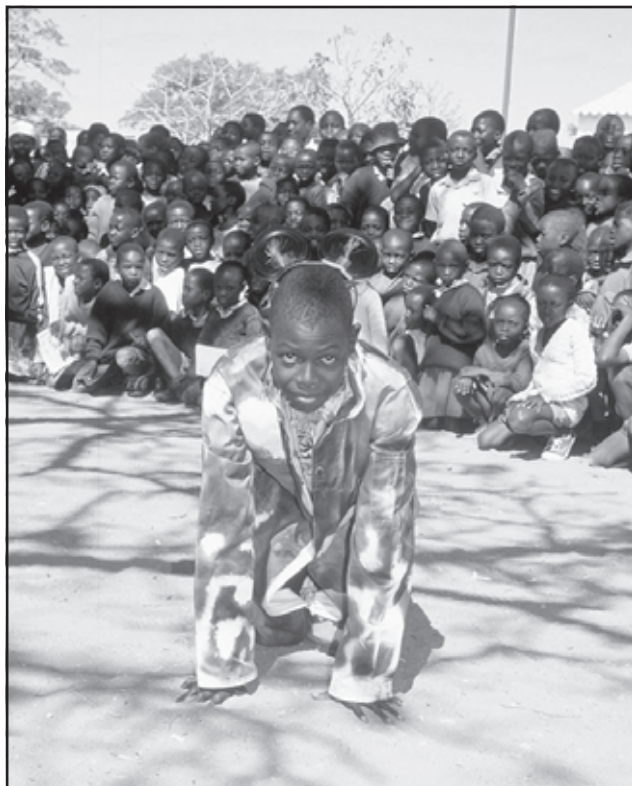
‘Education’ is an abstract concept difficult to define. There are numerous education theories, but there is no universally agreed or unchanging definition (Jarvis 1993). It is, therefore, often described through criteria used or through the processes of education. One example of defining education is that it equips the learner with knowledge, skills and experience at a level where insight and criticality are achieved. Peters (1966) argues that a person can be extremely knowledgeable or well informed, but this does not necessarily mean they are ‘educated’, they must also have some understanding of the ‘reason why’ of things. In other words, a level of cognitive perspective must be achieved. It is this cognitive perspective that conservation educators aim for when attempting to change attitudes or influence behaviour.

In order for sustainable education strategies to work in wild canid conservation, the different concepts, criteria and processes of education need to be understood. Changing the negative views of some people for canids will require a shift in attitudes at a societal level, which will involve ‘educating’ younger generations through the formal school system.

Below are brief descriptions of some of the terms used in education and communication:

- **Formal education** – the highly institutionalised, chronologically graded and hierarchically structured ‘education system’ spanning lower primary school and upper reaches of the university (Coombs and Ahmed 1974), often referred to as ‘front end education’. Formal education frequently refers to teaching method rather than to the structure of the educational provision (Jarvis 1993).
- **Non-formal education** – any organised systematic, educational activity carried on outside the framework of the formal system to provide selected types of learning to particular subgroups in the population (Coombs and Ahmed 1974). This structured education, delivered over a period of time would include community programmes, adult education programmes, and programmes run by voluntary organisations which are not primarily set up to provide education, but do so as part of their wider function.
- **Informal education** – generally defined as unorganised and often unsystematic, although it accounts for the great bulk of any person’s total lifetime learning (Jarvis 1993).

Young students engage in an African wild dog (*Lycaon pictus*) educational workshop, featuring costumes for role-playing exercises, such as the suit this young man demonstrates. Experiential learning techniques are used to illustrate behaviours and research methods, such as predator-prey chase simulations between groups of children, and exercises wherein the children become “hiding” wild dogs while the researcher (facilitator) uses tracking gear to find them. Painted Dog Conservation, Hwange, Zimbabwe, 2001.



Peter Blinston

- **Conservation education** – according to Orr (1994), all education is ‘environmental education’, but most conservation education programmes usually fall within the two categories of non-formal and informal education.

The World Conservation Union (IUCN) defines conservation education as: “a long-term process which seeks to modify the behaviour and attitudes of people by heightening their awareness of the natural environment and all its components.”

It should be borne in mind, however, that there are groups of people who are more aware of conservation and the natural environment, but they may not be disposed to protecting those parts of it that threaten their own livelihoods or welfare. Certain types of hunters and farmers may fall into this category, and may be resistant to any attempt to change or modify their behaviour.

- **Indoctrination** – the concept of indoctrination is particularly relevant in conservation education when dealing with different cultures and, therefore, different beliefs and value systems. Snook (1972) gives a simple definition: “The inculcation of a proposition or set of propositions in the pupil that are said to be certain when the teacher knows they are uncertain and with a lack of regard for any evidence to the contrary.”

Conservation educators need to be aware of the dangers of indoctrination by taking into account the beliefs, values and cultures of indigenous people. There may well be situations where we should ask ourselves whether it is morally or ethically right to impose our own values and beliefs, when implementing conservation strategies, on people in other societies? This may be particularly relevant when considering human-canid conflict mitigation, or where certain restrictions are placed on human activities in order to protect individual species or habitats. Conservationists may decide that hunters, trappers, ranchers or livestock owners pose the main threat to canids through their practices or behaviour. What is less straightforward is how to achieve canid conservation through education strategies without this leaning towards indoctrination, bearing in mind that attitudes toward canids are often entrenched in centuries of culture and tradition.

- **Communication** – the exchange and imparting of ideas and information, and the combination of effective sending and effective receiving. Communication as a conservation tool includes a wide range of activities, strategies, and approaches, of which some of these may be defined as being ‘educational’. In other words, effective communication brings about behaviour modification through a process of reflection and criticality. Others may simply impart information to raise awareness. When developing a conservation education programme, a whole range of education and communication strategies may be employed.

Table 18.1. Elements of success for conservation education programmes (Jacobson and McDuff 1997).

<p>Planning</p> <p>Have clear goals Identify measurable objectives Adopt an interdisciplinary approach Assess participants' social/education/economic backgrounds Ensure programme relevance to local populations Build necessary support (govt/community/industry) Maintain a budget plan Develop an organisational plan Plan for potential problems and resolution of conflicts</p>
<p>Implementation</p> <p>Follow an integrated approach Use existing organisations/groups Encourage active/voluntary participation Involve reluctant participants creatively Be sensitive to the audience Provide direct contact with the environment/resource Use key ecosystems/resources/species in programme Select appropriate educational media Use mass media Focus on economic/cultural values Provide conservation incentives Maintain informality/entertainment value of programme Be flexible</p>
<p>Evaluation</p> <p>Evaluate programme components/monitor programme Use more than one method of evaluation Collect feed back for programme modification/creation of new programmes Transfer programmes to local control and support Develop specific long-term plans for sustainability Disseminate programme results</p>

- Strategic action planning
- Research planning
- Operational planning
- Financial planning
- Evaluation and monitoring

Defining the conservation goal(s)

Primary conservation goals are usually straightforward – for example, to protect and conserve wild canid populations and/or their habitat. However, there may also be a number of secondary conservation goals, perhaps including the protection and conservation of prey species in the programme goals. Although it may be stating the obvious, it is a useful exercise to have the primary conservation goals as a starting point and to ensure that all programme goals are very clearly defined. Poorly defined goals will lead to problems when quantifying the criteria for success of the conservation goals, in terms of both the biological and social measures, and later when evaluating whether the criteria for success have been met. For long-term programmes, there will inevitably be a turnover in project staff. New staff joining a project will need to have a clear understanding of the conservation goals.

One of the main conservation problems for wild canids is human-canid conflict mitigation, especially in the case of humans and grey wolves (*Canis lupus*), where depredation of livestock is one of the major factors affecting wolves and conservation efforts. One of the key themes at the 2003 World Wolf Congress in Banff, Canada, was controlling wolf populations where predation on livestock and game species was perceived to be a problem. In this case, conflict mitigation may be the primary goal that underpins the education programme.

It is easy to see the contentious issues that arise, and the difficulties that may be encountered, in trying to develop an effective education programme to satisfy this goal when there is a polarisation of the different groups involved. Many hunters, ranchers and outfitters would like to see lethal control methods used, while conservationists favour non-lethal methods that would raise tolerance levels towards increasing wolf populations, such as livestock guardians, fencing, ‘scare’ devices and aversion conditioning. In such situations a number of primary and secondary conservation goals would be required.

Identifying threats and barriers to the conservation goal(s)

This is the key to solving the conservation problem or issue. In our modern world environmental problems are often anthropogenic:

- Habitat loss or fragmentation
- Human/carnivore conflict
- Predator-prey relationships – e.g., over-harvesting of prey species
- Political – legislation and policy

18.3 Developing a conservation education programme

Conservation education is extremely complex and varied, and there is no ‘one size fits all’ solution. However, there are a number of generic stages in developing an education programme, which, no matter how complex the conservation goals are, will apply to most programmes, and which can then be applied with some adaptation.

Jacobson and McDuff (1997) reviewed 15 successful conservation education programmes and identified 28 programme elements of success (Table 18.1).

The following sections take these ‘elements of success’ into account for the development and implementation of a conservation education programme focusing on any wild canid species.

18.3.1 Planning

Good planning will aid implementation and will also be a contributing factor in the overall success of the programme. Planning is required at every stage:

- Prevailing attitudes – often negative for wild canids
- Pollution
- Economics versus ethics – often there is a focus on ‘worth’ and ‘value’ in anthropocentric terms rather than biocentric terms

This list is not exhaustive, and a number of threats may occur simultaneously. Ethiopian wolves (*Canis simensis*), for example, are threatened by habitat loss and continued fragmentation, canid related diseases, overgrazing of highland pasture, human interference and persecution (due to livestock losses), hybridisation with domestic dogs, road kills, and accidental poisoning (which was aimed at other predators Williams 2003).

Simply identifying the threats and barriers is not enough. The exact nature of the threat needs to be clearly understood and the underlying reasons for the threat or barrier existing need to be clarified.

Note: In the case of grey wolves, the problem is exacerbated in countries where wolves have returned after a long absence, either through natural migration or recovery programmes. France provides such an example, since wolves and other large carnivores have been absent from the landscape for some time and the arrival of wild wolves from Italy in 1992 caused a great deal of hostility, especially from sheep farmers. Once wolves started to prey on sheep, farmers took to demonstrating with their flocks through the streets of Nice – an event that was also widely reported by the media throughout the world. Negative perceptions of the wolf have been ingrained over many centuries. Indeed, the stories such as Little Red Riding Hood and The Beast of Gevaudan originated in France. These ingrained attitudes, together with little or no experience of dealing with large predators, have left French farmers and local people ill-equipped to deal with the situation, and the result has been conflict.

The conservation problem and possible solutions also need to be placed into a wider geo-political and cultural context. Conservationists often carry out their work in other countries where there may be different languages spoken and a variety of different cultures. Identifying the threats and barriers to conservation (both direct and indirect), evaluating the underlying reasons and placing them in context will help when identifying and defining the target group(s) causing the problem.

Identifying and defining the target groups

For any conservation education programme to be successful in achieving both the conservation and education goals, it is vital that the target groups are clearly defined and understood. The ‘general public’ does not exist (Jacobson 1999), and the term ‘community’ is too general. A geographical community (town, city, neighbourhood) is usually defined by physical boundaries. Another concept of community is ‘community of interest’ where the members

share a common goal, purpose or interest. Furthermore, it may be necessary to divide target groups into sub-categories, depending on the education strategy being formulated and the ‘messages’ that need to be delivered.

In wild canid conservation, often the main threats are those groups who, for one reason or another, are in conflict with the aims of the conservationists. These may include:

- Hunters
- Farmers/livestock owners
- NGOs with conflicting agendas
- Local people

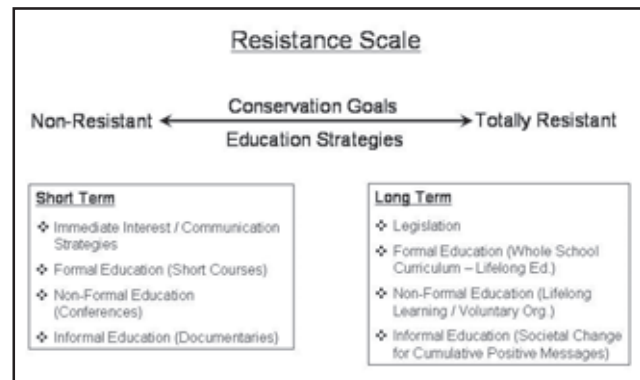
Aside from understanding the ‘who’ and ‘what’ of the conservation problem and its causes, the target group’s motivation(s) for their behaviour also needs to be clarified:

- economic survival
- livelihood protection
- physical protection
- competition for game or prey species
- attitudes and perceptions
- sport / recreational hunting

Understanding the target groups and what motivates their behaviour makes the task of effective education and/or conservation strategies much easier to apply. One method is to measure levels of resistance to the primary conservation goals. Target groups can be placed on a Resistance Scale (Figure 18.1) to help ascertain the quantity and level of ‘education’ needed, as well as the approximate time required to achieve the conservation goals. This is a very simple model, but it demonstrates the need to look very carefully at the target groups that the education programme is aimed at.

The more resistant the target group is to the primary conservation goal, the more difficult it will be to influence a change in behaviour that will help achieve the

Figure 18.1. Resistance Scale – a simple model to identify levels of resistance to primary conservation and education goals in order to aid the implementation of appropriate education strategies.



conservation goals. At the lower (non-resistant) end of the Resistance Scale there are those groups who may simply be unaware of the conservation problem and who are, on the whole, in favour of conservation. Raising awareness of the problem through communication and public relations is more likely to influence a change in perception, attitude and behaviour. However, it is these groups who are least likely to have their lifestyles affected by the conservation problems or issues and who will find it easier to support a conservation programme. In Norway, for example, local people were initially very negative towards the natural reintroduction of wolves into the area. In this case, all that was required to change negative perceptions was to increase local knowledge of large predators and ecology (see Box 18.1).

Groups who are totally resistant to the conservation goals, however, will have no motive for modifying their behaviour. In some human-canid conflict situations, certain target groups may be openly hostile to conservationists. A farmer or hunter whose very survival may depend on killing wild canids is unlikely to change this behaviour unless there is an incentive or disincentive to do so. People tend to make decisions that maximise their self-interest in the short term (Norgaard 1997).

Wild canids have the additional barrier of negative perceptions and attitudes. In a study of public perceptions of predators, particularly the wolf and coyote, Kellert (1985) found that livestock producers expressed very

negative attitudes toward the wolf, although wolves constitute a negligible factor in livestock predation.

Note: It must be borne in mind, however, that farmers and hunters do not fit neatly into homogenous target groups when applying conservation strategies. There are some farmers and hunters who are not intolerant of canids, and who actively engage in their conservation. For example, the Thirteen Mile Lamb and Wool Company in Montana, USA has adopted a 'predator friendly' policy and practises non-lethal methods of predator control using llamas to protect livestock. Thirteen Mile produces knitwear from its sheep's wool depicting wolves which they sell to supplement their income, even though their livestock have been threatened by wolves.

It is clear that the problem at the higher end of the Resistance Scale becomes much more complex, and can have a knock-on effect. Hunting and farming groups also have a powerful political voice in some countries and may influence policy-makers. In these situations, other conservation strategies are needed, which may require some form of education for other groups of people in order to ultimately achieve the conservation goals. This comes back to the question of whom needs educating and why? Awareness of such complexities, however, will help formulate education strategies which can be used in conjunction with other conservation strategies, and will hopefully promote some form of unity and cohesion in working towards the overall goals of the project.

Box 18.1. From fear to curiosity – a conservation education programme in Norway.

In 1997, a grey wolf pack established itself in southern Norway. The presence of wolves, in an area where they had been absent for many years, provoked an acute fear and insecurity among the local people. In November 2000, the Ministry of the Environment and the local municipality started a two-year, non-formal education and communication programme in the Stor-Elvdal community, which has 3,000 inhabitants spread throughout the countryside. The programme comprised of carnivore news bulletins being sent to every household six times *per annum*, together with a range of outdoor education activities delivered through local schools for 500 children, for 30 days each semester. Activities included radio-tracking, snow-tracking, howling events, investigating carcasses and dens. A website for the project was also developed (<http://www.rovdyrkunnskap.net>) which attracted children from all over the country who wanted to learn more about the programme.

At the start of the project, there was a negative reaction from the community. However, as soon as the children became involved in the activities, attitudes started to change. The children were enthusiastic about the discoveries they were making and the new knowledge they were acquiring. This enthusiasm was then generated on to the parents, who then expressed interest and a willingness to learn more about the activities. A range of 20 activities was then arranged for the adults, with between 30 to 150 participants in each event. The aim of the education programme was to increase both theoretical and practical knowledge through 'self-experience'.

After two years, the initial evaluations show that the programme has achieved some success, with local people demonstrating greater curiosity about wolves and other large carnivores, and the fear of wolves has been greatly reduced. People have subsequently asked for more information and further activities related to large carnivores, and have even become involved in population estimation research on wolves in the area.

For further reading about the programme (only available in Norwegian at present), please see: NOVA Rapport 9:2003 Evaluering av "Prosjekt rovdyrkunnskap" i Stor-Elvdal kommune by Mette Svenningsen & Ketil Skogen (report available from <http://www.nova.no>)

Further information about the project may be obtained from:
Kristin Evensen Gangaas
The Ministry of the Environment
Stor-Elvdal kommune
2480 Koppang
Norway
Email: kristin.evensen.gangaas@stor-elvdal.kommune.no

Defining the education aims and objectives

The education goals will usually be aimed at modifying the behaviour of target groups in order to achieve the overall conservation goals. Jacobson (1997) suggests several goals of conservation education, including:

- Increasing public knowledge and consequent support for the development of appropriate environmental management and conservation policies.
- Fostering a conservation ethic that will enable responsible natural resource stewardship
- Altering patterns of natural resource consumption.
- Enhancing the technical capabilities of natural resource managers.
- Incorporating resource management concerns into private sector and government policy-making processes.

The reasons for and type of education (formal, informal, and non-formal or simply a public relations or communications programme) needed by certain groups and sub-groups need to be ascertained. Some groups may simply need to be made aware of a particular problem to change their perceptions, others may require attitude or behaviour changes. The first is less problematic. There are fewer ethical and philosophical considerations. Providing information and raising awareness may not require the recipients to take action or to make any radical changes to their lifestyle. The degree of behaviour modification will depend on the degree of resistance to the primary conservation goal and will determine the nature of the education programme. Education aimed at changing attitudes and behaviours, on the other hand, requires a great deal of reflection and consideration of the complexities involved. Canid biologists and conservationists are well aware of the hostility faced when the species they are trying to conserve come into direct conflict with human activities, such as hunting, livestock farming, etc. At this level, a number of conservation strategies may be required, e.g., education, legislation, and incentives or disincentives. In Bulgaria, for example, a campaign that provides breeding pairs of livestock guarding dogs to local shepherds has been combined with a number of non-formal education and communication initiatives. These include courses provided at local schools, widely distributed brochures, posters, photo exhibitions, calendars, and media coverage (working with local television). Legal action was also taken to lobby for changes in hunting laws, which was successful (Tsingarska-Sedefcheva 2003).

Education strategies as a conservation tool should not be used in isolation. If behaviour modification is required at a societal level, and a variety of strategies are to be applied, then this will require a multidisciplinary approach that might also require a change in attitude or behaviour on the part of biologists and conservationists.

18.3.2 Implementing a strategic action plan

A strategic action plan will provide an overview of what needs to be achieved, and will allow benchmarks to be incorporated to aid the monitoring and evaluation process (see Table 18.2).

Table 18.2. Some suggestions on headings for the strategic action plan are given below, although these may vary from programme to programme.

- | |
|---|
| <ul style="list-style-type: none"> • Background Information/Introduction <ul style="list-style-type: none"> — Define the “problem” |
| <ul style="list-style-type: none"> • Aims and Objectives of the Programme <ul style="list-style-type: none"> — Primary Conservation Goal(s) <ul style="list-style-type: none"> • Short, Medium, Long Term — Education Aims and Objectives <ul style="list-style-type: none"> • Short, Medium, Long Term |
| <ul style="list-style-type: none"> • Research <ul style="list-style-type: none"> — Target Group Research <ul style="list-style-type: none"> — Methodologies — Research Results |
| <ul style="list-style-type: none"> • Conservation Education Strategy/Plan <ul style="list-style-type: none"> — Define the Target Groups — Define Methods of Education to be Used <ul style="list-style-type: none"> • Formal Education – school curricula • Non Formal Education – adult education, structured programmes • Informal Education <ul style="list-style-type: none"> — NGOs — Community Education — Communication/Awareness Programmes — Mass Media |
| <ul style="list-style-type: none"> • Resources Required <ul style="list-style-type: none"> — Human Resources <ul style="list-style-type: none"> • Project Leaders/Directors • Educators/Trainers • Administrative Staff • Volunteers • Researchers — Financial — Capital Equipment — Operational <ul style="list-style-type: none"> • Premises – for project staff • Locations for delivery of education (where applicable) • Vehicles • Operational Systems and Procedures |
| <ul style="list-style-type: none"> • Legal Issues <ul style="list-style-type: none"> — Conventions and Treaties — Licences and Permits — Local, National and International Legislation — Employment Law — Health and Safety Legislation — Company/Organisation Structure – legal entity |
| <ul style="list-style-type: none"> • Financial Plan <ul style="list-style-type: none"> — Budgets — Income and Expenditure Forecasts — Cashflow Forecasts — Balance Sheets — Management Accounts/Internal Financial Information |

18.3.3 Evaluating the education programme

Evaluation is an important stage in a conservation education programme, and yet constraints on resources (financial, available staff and time) may lead to this part of the process being given less attention. In an analysis of successful wildlife conservation education programmes, Pomerantz and Blanchard (1992) reveal that few of the case studies found in a literature search included an evaluation component. Similarly, of the 56 reports included in Norris and Jacobson's (1998) analysis, fewer than half reported using some type of evaluation. However, in the same analysis, there were significantly higher rates of success for the programmes that did include evaluation.

Kleiman *et al.* (2000) describe three important aspects of effective evaluation:

1. Having an individual with leadership ability and considerable expertise to organise the format and oversee the review process;
2. Ensuring, at the outset, that there is agreement among the programme participants and the review committee on the goals and objectives of the conservation programme, what is to be evaluated, and the criteria for defining success; and
3. Ensuring that the programme is inclusive and involves all participants and stakeholders.

When incorporating evaluation into a programme consideration needs to be given to the following key elements:

- Measuring success – what the programme has accomplished. Have the criteria been met for both the conservation and education goals?
- How effectively has the programme functioned as a process?
- How are changes going to be made to the programme as a result of evaluation findings?
- How will information obtained from the evaluation be disseminated?

Evaluation should be integrated into every conservation programme, and it is important that it is included as part of the process when planning for resources. Otherwise it is difficult to evaluate the impact of a programme, as well as the effective features of successful conservation education programmes (Jacobson and McDuff 1997).

18.4 Resources required

18.4.1 Human resources

Human resources will probably be multi-agency and involve several stakeholders such as scientists, researchers,

educators and trainers, conservationists, funders and sponsors, administrators, volunteers, policy-makers, community groups, local businesses, NGOs, and the media. The various roles will need to be clearly defined, and good communication between the various agencies or stakeholders is extremely important.

Effective project leaders are vital to the success of the programme and will need to be identified and accepted as they will be the driving force behind the project. Strong, effective leaders will also help to deal with any conflict resolution that may potentially threaten the programme.

18.4.2 Financial resources

Obtaining the necessary funding for any conservation project requires a great deal of time and energy. For conservationists this is a part of the process that is often the least welcome, and one that few people are trained to undertake. Financing a project is often seen as a 'necessary evil' and for this reason can often be undertaken half-heartedly. It is also a continuous process, especially if the project is long term.

Wherever possible, a professional fundraiser should be recruited – someone who has the necessary expertise in approaching potential funders and sponsors and who can prepare the necessary documentation, grant proposals and funding bids. Some fundraisers may offer to help on a voluntary basis until the project is more established. Others operate on a 'commission' basis whereby if they obtain the funding the project requires, they are paid a percentage.

Prepare a detailed financial plan. This can be done using spreadsheets or simple accounting software packages.

The first step of this process is to establish how much finance will be required and what this will be for:

- Salaries and staff expenses
- Capital equipment
- Premises
- Educational materials
- Research expenses
- Travel expenses
- Operational expenses and other overheads

Accessing funding can be a long, difficult and dispiriting process. However, even a cursory internet search will demonstrate that there is a lot of money available for conservation projects in the form of grants, awards and prizes, and sponsorships. Some conservation projects may also attract legacies and bequests, and donations. The key is to match your programme objectives to the funders' criteria. Be creative, and don't forget that there are two angles to conservation education programmes – the conservation angle and the education angle. There is funding available for both.

18.5 Conclusions

Education has an important role in conserving canids. This includes changing perceptions and attitudes in order to increase tolerance and foster co-existence. A recurring theme is conflict mitigation and addressing the issues that arise as a result of human/canid conflict – livestock depredation, competition for game species, negative perceptions and attitudes that provoke a ‘fear’ response, etc. It should also be recognised, however, that the issues involved are extremely complex and varied. Trying to quantify conservation education raises far more questions than it answers. But understanding the concepts of education, and how education can be applied in conjunction with a number of other conservation strategies will help to make the task of developing a conservation education programme much easier.

At present, there is still very little practical conservation education information available. There is also a lack of cohesive information with regard to conservation education programmes that are taking place worldwide. There are many excellent and successful initiatives, and

we can also learn lessons from the ones that are not so successful. This information needs to be made more widely available. There needs to be a lot more cohesion with regard to conservation education that cuts across all boundaries and promotes a sense of unity and common purpose. If the target groups requiring the education are the policy-makers, then conservationists (in the broadest sense – encompassing everyone who works for the protection of species and habitats) need to have a stronger voice to be able to lobby at the highest levels.

Conservation education should be developed with a long-term view. Some of the attitudinal shifts and behavioural changes required are at societal level. This will mean implementing a combination of strategies, starting with early years education through the formal school system right through to adult education through non-formal, informal and communication programmes.

And finally, it is important to understand the problems and the target groups in great depth. It is this detailed knowledge and understanding that should underpin any conservation education programme.