Conserving wild dogs (*Lycaon pictus*) outside state protected areas in South Africa: ecological, sociological and economic determinants of success

by

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Summary

The restricted geographic range and tenuous conservation status of wild dogs in South Africa were the motivating factors behind this study. Wild dogs have been extirpated from most of their historic range in South Africa, and now occur in three limited distributions: a) one viable population in Kruger National Park; b) a protected meta-population, consisting of 11 packs in six sub-populations (four on state owned reserves - Hluhluwe-Umfolozi Park, Madikwe Game Reserve, Marakele National Park, Pilanesberg National Park, and two on privately owned reserves - Karongwe Game Reserve, and Venetia Limpopo Nature Reserve), and; c) ~ 76 unprotected individuals in 17 packs and dispersing groups occurring outside protected areas, primarily in the game ranching areas of the extreme north and north east.

Prior to the establishment of the proposed transfontier parks, the best prospects for range expansion likely exist on private land. My study investigated some of the ecological, sociological and economic issues associated with wild dog conservation on private land under various scenarios.

Over the last few years, the focus of conservation efforts and donor funding expenditure (72.6% of funding) has been the establishment of the meta-population. This been effective - the target size (nine packs) of the meta-population has been exceeded in six years, four years less than the targeted schedule (10 years). From here, there are two ways in which donor funding might be used to achieve further range expansion outside state protected areas, through expansion of the meta-population by reintroducing wild dogs onto private nature reserves, and through the conservation of wild dogs *in situ* on ranchland. For either strategy, an estimated minimum area of 158.5 km^2 is required to support the predation requirements of a pack of 12 wild dogs in northern South Africa, 172.8 km^2 in eastern South Africa, and 354.2 km^2 in northeastern South Africa.

Private reserve owners may not be willing to accept the costs of predation by wild dogs in the absence of compensation. Compensation for predation ($9,563 - $101,762 / year), in addition to the high start up costs of wild dog reintroductions ($36,880) would increase annual donor funding requirements by 1.3 - 4 times, and reduce the cost efficiency of this strategy below that of alternative conservation options. However, there is potential to generate substantial revenue from wild dog-based ecotourism ($11,000 - $60,000 / pack / year), and given careful reserve selection, tourism benefits can exceed the costs.

Consequently, private reserve owners might be encouraged to reintroduce wild dogs at their own cost. In line with this, the Wild dog Advisory Group-SA has received enquiries from several private reserve owners interested in reintroducing wild dogs onto their properties. The expansion of the meta-population should be limited to state-owned reserves and private reserves willing to carry the costs.

There are more wild dogs occurring outside protected areas than previously recognised. Potentially important founder populations occur in game ranching areas in eastern (1 – 3 resident packs and dispersing groups), northern (1 – 5 resident packs and dispersing groups) and western Limpopo (1 – 5 resident packs and dispersing groups), and large areas (88,750 km^2) of potentially suitable habitat for range expansion are currently available. Persecution by landowners remains a significant problem, however, and until this is controlled, range expansion is unlikely to occur. Negative attitudes (47.7% of ranchers) are typically based upon perceived or real economic costs associated with wild dogs, and the removal of cost burdens from landowners is the most direct way in which attitudes might be improved. Despite the high annual costs associated with predation by wild dogs on ranchland (\$11,942 - \$115,761), the low logistical costs (\$3,572 initially, and then \$15,382 annually thereafter) associated with conserving wild dogs *in situ* on ranchland render this option more cost efficient than the reintroduction of wild dogs onto private reserves (14 – 27 packs conserved / \$100,000 cf. 3 – 19 packs / \$100,000). Furthermore, tourism revenue from wild dogs has the potential to offset the costs of their predation on ranchland under most scenarios, and promoting the conservation of wild dogs *in situ* on ranchland by assisting ranchers in establishing wild dog-ecotourism operations should be the focus of future conservation efforts. A substantial proportion of ranchers (52.3%) are positive towards wild dogs, and private landowners are potentially important facilitators in the conservation of the species in South Africa.

The focus of future conservation efforts involving wild dogs in South Africa should be to establish wild dog populations in the proposed Limpopo / Shashi and Lubombo transfrontier conservation areas as soon as they are established, to encourage private reserve owners to reintroduce wild dogs at their own expense, and to promote the conservation of naturally occurring wild dogs *in situ* on ranchland, by encouraging and assisting ranchers to establish wild dog-ecotourism programmes.

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CHAPTER 1

Conserving large carnivores outside state protected areas: an introduction

Large carnivores are predisposed to conflict with humans and are consequently more difficult to conserve than most other taxonomic groups (Linnell et al. 2001). In recent history, humans have been responsible for the extinction of some large carnivore species (e.g. Falkland Island wolf *Dusicyon australis*), and for substantial reductions in the distribution of many other species (e.g. brown bears *Ursus arctos*, lions *Panthera leo*, jaguars *Panthera onca,* and wolves *Canis lupus,* Johnson et al. 2001). Anthropogenic mortality continues to constitute the most significant threat to the persistence of many carnivore species (Woodroffe & Ginsberg 1998). Despite this, there are some positive trends in large carnivore conservation, particularly in North America and Europe. A combination of increased legal protection, limits to the use of poisons, expanding forests, recovering prey populations, and reintroduction programmes have resulted in increasing populations of wolves and cougars *Felis concolor* in the USA, and brown bears, wolves, and lynx *Lynx lynx* in parts of Europe (Breitenmoser 1998; Swenson et al. 1998; Linnell et al. 2001). In most other parts of the world, however, rapid expansion of human populations coupled with ineffective regulation of hunting is linked to continuing population declines and local extinction of large carnivores (Woodroffe 2000; Linnell et al. 2001).

1.1 Reasons for large carnivore / human conflict, and the difficulty associated with conserving remaining populations

1.1.1 Ranging behaviour and life history characteristics

Due to their position at the top of the food chain, large carnivores require large areas to exist (Macdonald & Sillero-Zubiri 2002). An estimated 10,000 kg of prey are required to support 90 kg of carnivore, and large predators are invariably much rarer than their prey (Carbone & Gittleman 2002). Large area requirements reduce the number of protected areas, or habitat fragments outside protected areas capable of effectively conserving large carnivore species. Species such as brown bears and wild dogs *Lycaon pictus* pose particularly acute problems for conservation as a result of their tendency to range far beyond the borders of protected areas (Woodroffe & Ginsberg 1998). Large carnivore species are also often characterized by a K-selected life history pattern, with delayed reproductive maturity and small litter sizes, reducing their capacity to tolerate persecution (Ferguson & Lariviere 2002).

1.1.2 Competition with humans for prey

Predation upon livestock is a source of conflict between large carnivores and humans virtually wherever they coexist - for example cougars killing goats in Argentina (Johnson et al. 2001), lions killing cattle and sheep in African savannas (e.g. Parry & Campbell 1992), and lynx killing sheep in the Swiss Alps (Breitenmoser 1998). The economic impact of predation by large carnivores upon livestock is variable. In some instances the impact is very high - for example predation by snow leopards *Panthera uncia* in Nepal results in losses equal in value to a quarter of the average per capita income for affected households (Oli et al. 1994). In other instances, the impact of large carnivores is small relative to other causes – for example on a ranch in Kenya, carnivores killed 2.2% of sheep, compared to 7.8% killed by disease (Mizutani 1993). The impact of predators upon livestock is rarely evenly distributed among community members, and some farmers may suffer catastrophic losses, while the majority lose nothing (Archabald & Naughton-Treves 2001). In Gokwe communal land in Zimbabwe, for example, although the average annual cost per household due to predation upon livestock is US$ 13 per year, certain households suffer much greater losses due to occasional multiple killings of cattle by lions (Butler 2000). Carnivores also compete with humans for wild prey. Local hunters in the Canton Valais in Switzerland are opposed to the return of the wolf, because they perceive it as a competitor for the wild ungulates they hunt (Glenz et al. 2001). Conflict between humans and large carnivores due to losses of livestock and wildlife is often exacerbated by a lack of accurate information on the extent of, and true causes of mortality, and losses are often exaggerated. For example, wild dogs were responsible for only 2% of cattle losses in northwest Zimbabwe, contrary to exaggerated claims by ranchers (Rasmussen 1999).

1.1.3 Predation on humans

Globally, large carnivores cause hundreds of human fatalities every year, creating a highly emotive conflict scenario (Linnell 2002). Relative to the number of people coexisting with large carnivores however, the number of human deaths is remarkably low (Chakrabarty 1992). Attacks on humans typically result from injury to individual carnivores reducing their ability to catch natural prey, habituation and loss of fear of humans, the defence of kills by carnivores from prospective thieves, or the occurrence of 'problem animals' (Treves & Naughton 1999). An additional problem is the increasing encroachment by humans into wildlife areas and increasing contact between large carnivores and people as a result. For example, Yamazaki & Bwalya (1999) reported three fatal attacks by lions on people living in a wildlife management area in Zambia. Conversely, in North and South America, expanding populations of cougars have resulted in increasing numbers of attacks on humans (Johnson et al. 2001). Most human fatalities are caused by large felids, and most notably tigers *Panthera tigris* (Sillero-Zubiri & Laurenson 2001). In the Sundarbans mangrove forests in India, for example, between 36 and 100 people are killed annually by tigers (Chakrabarty 1992).

1.1.4 Exploitation of commercially valuable carnivore species

Historically, the exploitation of commercially valuable species has resulted in conservation problems for a variety of large mammals, including black *Diceros bicornis*, and white rhinoceros *Ceratotherium simum*, elephants *Loxodonta africana*, and several pinniped species. For example, northern elephant seals *Mirounga angustirostris* were over-exploited for their blubber in the 19^{th} century, and reduced to a population size of 10 - 20 individuals as a result (Johnson et al. 2001). The fur trade is a problem specific to carnivores, and over exploitation has resulted in conservation problems for several species. For example, in the 1960s, 15,000 jaguar, and 80,000 ocelot *Leopardus pardalis* pelts were removed annually from the Amazon, resulting in population declines (Smith 1976). Changing fashions and increased legislation have, however, reduced exploitation levels and by the late 1980s jaguar pelt prices had dropped by ~ 95% (Swank & Teer 1987). Commercial exploitation continues to be a problem for some species, as evidenced by the dramatic increase in tiger poaching to meet the growing demand for tiger bones and organs for traditional Asian medicine (Karanth & Stith 1999).

1.1.5 Human prejudice

Human perceptions and attitudes play an important role in determining which species are tolerated in a given area. For example, Kenyan ranchers tolerate large felids while persecuting spotted hyaenas *Crocuta crocuta,* despite the fact that the felids kill more livestock (Frank & Woodroffe 2001). Canids, in particular, appear to be the victims of irrational human prejudice. In North America, while bears are typically popular, wolves and coyotes *Canis latrans* are among the least liked of animals (Kellert 1985), perhaps due to negative childhood stories (Berg 2001). In several European languages, wolves are associated with negative expressions associated with danger and brutality (Dingwall 2001). Likewise, African wild dogs fare poorly in the public eye relative to other large carnivores, due to a reputation as wanton killers (Fanshawe et al. 1991). Overcoming human prejudices represents a major challenge in promoting coexistence between people and predators, and much research has been done on assessing the bases of attitudes towards large carnivores (e.g. Berg 2001; Dingwall 2001; Zimmerman 2001).

1.2 Approaches to the conservation of large carnivores

The best solution for large carnivore conservation is proclamation and maintenance of large protected areas (Terborgh & van Schaik 2002). However, due to increasing human populations and increasing competition for space, the scope for creating new protected areas is dwindling. Furthermore, few protected areas are large enough to permit a full range of ecosystem processes, or adequate protection for viable populations of large carnivores (Miller et al. 1999). For example, lions ranging from the Kgalagadi Transfrontier Park in Botswana / South Africa are regularly persecuted by ranchers in response to occasional cattle losses (Castley et al. 2002). Existing parks systems currently leave 93% of the world's surface unprotected and many parks, particularly in the developing world, are under-protected 'paper parks' (Langholz et al. 2000). African protected areas are becoming increasingly insularised and wildlife reserves need to be substantially larger than those presently established to preserve current species assemblages (Burkey 1995). In light of these shortcomings, there is an increasing move towards conservation approaches based upon creation of incentives aimed at encouraging local people to coexist with wildlife outside of parks (Prins & Grootenhuis 2000). Conservation efforts outside protected areas increase the area available to wildlife, potentially create buffers for populations occurring within parks, and potentially protect habitat types not represented within parks systems. For example, vital habitat for the San Joaquin kit fox *Vulpes macrotis mutica* occurs on farmland in California (Innes et al. 1998). Conservation outside protected areas is difficult, however, and many of the problems associated with conserving large carnivores are exacerbated.

1.2.1 Ecological approaches

Conserving carnivores outside protected areas can be achieved *in situ* by protecting intact habitat patches, through augmentation of existing populations by releasing additional animals, or through the reintroduction of species into areas from which they have been removed as a result of human activity. For each of these methods, the design of an effective conservation strategy is dependent upon an understanding of the behavioural ecology of a given species. For example, the area requirements for effective conservation vary between species. Extinction is predicted for grizzly bears in reserves smaller than 3,981 km^2, whereas the critical reserve size for the persistence of black bears *Ursus americanus* is predicted to be as small as 36 km^2, due partly to differences in ranging behaviour (Woodroffe & Ginsberg 1998). Reintroduction programmes are often severely affected by post-release ranging by large carnivores, and release sites must be far enough away from source populations to prevent reintroduced animals returning to the site of capture (Linnell et al. 1997).

Life history and behavioural ecology also determine the amount and type of prey required to sustain carnivore populations, and affect the extent to which species can adapt to habitats modified by human activity. For example, whereas lynx are strictly carnivorous and cannot survive in the absence of wild ungulate prey, bears and wolves are more able to survive on alternative food sources such as human waste products (Meriggi & Lovari 1996). The presence of sufficient suitable wild prey in an area also affects the chances of conservation efforts involving a given carnivore species succeeding, by reducing roaming beyond the borders of conservation areas, and reducing livestock depredation (Miller et al 1999).

Finally, life history traits are important in determining sensitivity to adult and juvenile mortality, ability to tolerate habitat modification, and the size of founder populations required for reintroductions. For example, conservation strategies for species such as neotropical cats, which are short lived and produce few large neonates following lengthy gestation, should be focused upon juvenile survival, whereas conservation strategies for K-selected competitors such as pinnipeds, should focus upon reducing adult mortality (Ferguson & Lariviere 2002). Variation in life history traits leads to a corresponding variation in necessary conservation strategies, ranging from complete protection and active management (e.g. the black footed ferret *Mustela nigripes*), to protection in conjunction with limited control of problem animals (e.g. tigers), to sustainable harvest of species with sufficient numbers, growth rates and economic value (e.g. cougars), and finally, to the control of species with no economic value (e.g. coyotes, Johnson et al. 2001).

1.2.2 Sociological approaches

The conservation predicament of most large carnivores is directly attributable to conflict with humans (Woodroffe 2000), and consequently involving communities in the development of conservation strategies is imperative. Socio-political and economic considerations become more important as carnivore conservation is extended beyond protected areas, as conflict with humans increases (Mech 1998; Glenz et al. 2001). In Europe, while expanding carnivore populations are frequently supported by urban populations, the views of people directly affected by carnivores are often less positive (Breitenmoser 1998; Ericsson & Heberlein 2003). A first step in conserving predators outside protected areas is to work with the communities most affected to determine the source and extent of conflict, and to identify ways in which conflict might be reduced.

Experience from Norway suggests that conflict between people and expanding large carnivore populations could have been reduced by implementing information campaigns prior to and during natural re-colonisation by large carnivores (Zimmerman et al. 2001). Effective public relations and education campaigns are also a vital prerequisite for carnivore reintroduction programmes (Reading et al. 1997). Outreach work done prior to and during the reintroduction of wild dogs into Venetia Limpopo Nature Reserve in South Africa, for example, yielded promises from ranchers not to shoot dogs if they escape from the reserve (Davies, pers. comm.). In some cases, attitudes towards conservation can be improved simply by acknowledging that people are adversely affected by the presence of large carnivores (Sillero-Zubiri & Laurenson 2001). In other cases, however, intensive educational programmes are required – as illustrated by an example from Namibia, where 95% of ranchers interviewed indicated that they had no knowledge of the conservation predicament of cheetahs *Acinonyx jubatus* (Marker-Kraus & Kraus 1997). However educating people with entrenched attitudes towards large carnivores is difficult (Ericsson & Heberlein 2003), and in intensive conflict scenarios, education and awareness campaigns are unlikely to succeed without addressing the economic bases of conflict.

1.2.3 Economic approaches

The emerging discipline of ecological economics has a vital role in strengthening the case for conservation, by highlighting the economic value of wildlife, and identifying the economic pressures threatening species (Edwards & Abavardi 1998). Recent studies have identified three related economic reasons for continued habitat conversion and species losses: a) the lack of information on the true value of ecosystem goods and services; b) the failure of markets to capture the benefits provided by nature to human welfare, and c) the use of perverse subsidies by governments to promote environmentally harmful agricultural activities (Constanza et al. 1998; James et al. 1999; Balmford et al. 2002). For example, unprofitable sheep farming in the Swiss Alps is heavily subsidised, creating conditions conducive to conflict with large carnivores (Breitenmoser 1998). Ultimately, wildlife will only persist in the long term if the value of its conservation outweighs these costs, and out-competes alternative land uses that contribute to human welfare (Child 1995). It is imperative that conservation strategies are designed to reduce costs incurred by local communities, and to promote exploitation of the use values of large carnivores that can be captured by conventional markets, to create financial incentives for conservation. Attempts to reduce the costs associated with the conservation of large carnivores have included the following:

a) Improved livestock husbandry. For example, livestock losses in Kenya can be reduced by constructing bomas strong enough to prevent cattle from panicking and breaking out when they become aware of the presence of lions outside (Frank & Woodroffe 2002).

b) Problem animal control. Livestock losses are sometimes attributable to single individuals, in which case an appropriate solution is the removal of that individual. For example, some British gamekeepers selectively remove individual foxes *Vulpes vulpes* responsible for losses of lambs (Sillero-Zubiri & Laurenson 2001).

c) Compensation schemes. Compensation schemes aim to promote coexistence between large carnivores and people through the reimbursement of livestock losses. For example, 12 states in the USA have compensation schemes for bears, five states for wolves, and four states for cougars (Wagner et al. 1997).

d) Removing perverse subsidies. The removal of price distortions which previously disadvantaged wildlife relative to livestock-based land uses has resulted in a proliferation of wildlife in southern Africa, in the form of game ranching on private land, and community based wildlife management schemes on communal land (Child 2000). These land use changes have had positive benefits for large carnivores in some areas – for example, in southeastern Zimbabwe, game ranching has led to population increases of cheetahs and wild dogs (Pole 1999).

Potential benefits associated with conserving large predators can be categorised as consumptive or non consumptive:

a) Consumptive utilisation involves hunting for sport or for animal products, or capture and live sale. The trophy fee for lions in Africa ranges from \$3,000 - \$15,000 (Loveridge & Macdonald 2002), and large carnivores are critically important for the trophy hunting industry in Africa (e.g. Creel & Creel 1997). The CAMPFIRE and ADMADE programmes in Zimbabwe and Zambia represent attempts to promote conservation outside protected areas through the generation of income from safari hunting (Balakrishnan & Ndhlovu 1992; Child 1996).

b) The most widespread form of non-consumptive utilisation is ecotourism. Global ecotourism, defined as "tourism compatible with conservation and posing minimum threat to local culture and society", is growing at rates of up to 15% per year, and has increasing potential to contribute to safeguarding biodiversity (Gossling 1999). Visitors to protected areas are willing to pay substantial sums to view certain wildlife species – in Kenya, for example, the net returns through ecotourism from the wildebeest *Connochaetes taurinus* migration are estimated at US$ 125 - 150 / animal / year (Earnshaw & Emerton 2000). Carnivores are especially popular among tourists (Davies 1998), and exploitation of their tourism value outside protected areas has potential to offset costs and promote coexistence between people and carnivores. The International Wolf Centre in Minnesota, for example generates an estimated US$ 3 million annually for the local economy (Mech 1998).

1.3 Carnivore conservation outside state protected areas: wild dogs in South Africa as a case study

The conservation predicament of wild dogs in South Africa is the result of a variety of ecological, sociological and economic factors. Consequently, wild dogs represent a useful model species with which to investigate the determinants of success in the conservation of a large carnivore species outside state protected areas.

1.3.1 Current conservation status of wild dogs

Historically, wild dogs occurred throughout sub-Saharan Africa, with the exception of true rainforest and desert (Creel & Creel 2002). There has been a dramatic reduction in numbers and geographic range over the last 30 years, with latest estimates suggesting that as few as 3000 - 5500 individuals remain (Fanshawe et al. 1997). Wild dogs have fared especially poorly in north and west Africa (Fanshawe et al. 1997) and viable populations are now restricted to southern (Botswana, Namibia, South Africa, and Zimbabwe), central (Zambia), and east Africa (Tanzania).

The ecological requirements of wild dogs predispose them, arguably more than most large carnivore species, to conservation difficulties. Large area requirements and naturally low population densities are the basis for their conservation predicament (Creel & Creel 2002). Wild dogs inhabit larger home ranges than expected for their body size (Gittleman & Harvey 1982), and utilise larger areas than other canids, or ecologically similar African carnivores (Creel & Creel 2002). Mean annual home range areas vary from 379 km^2 in Selous (Creel & Creel 2002) to 2,460 km^2 in the Serengeti (Burrows 1992). Correspondingly, wild dogs occur at densities much lower than those of competing carnivore species (Woodroffe et al. 1997a), ranging from a maximum of 4 dogs / 100 km^2 in Aitong in Kenya (Fuller & Kat 1990) to a minimum of 0.67 dogs / 100 km^2 in Serengeti (Creel & Creel 1996). The largest protected areas contain relatively few wild dogs, for example the 43,600 km^2 Selous Game Reserve contains an estimated 880 adult wild dogs (Creel & Creel 2002), and most protected areas contain much smaller populations (Woodroffe & Ginsberg 1999). Furthermore, ranging behaviour renders wild dogs highly susceptible to edge effects, and local extinction is predicted in reserves smaller than ~ 3,600 km^2 (Woodroffe & Ginsberg 1998).

Wild dogs have an obligatorily cooperative breeding system, which results in inverse density dependence, and renders them highly sensitive to adult mortality (Courchamp et al. 2000). Correspondingly, the threshold human density above which wild dog extinction is predicted (0.7 - 6.3 people / km^2) is far lower than that for most other carnivore species (Woodroffe 2000). Persecution by humans, in conjunction with habitat loss, is the most important reason for the decline in numbers of African wild dogs (Woodroffe & Ginsberg 1999). Persecution appears to be based upon both perceived and real economic costs, and human prejudice. Wildlife managers actively persecuted wild dogs in several countries throughout much of the 20th century (Creel & Creel 2002), and today, an estimated 27% of mortality in protected areas is attributable to human persecution (Woodroffe et al. 1997b). Large areas of natural habitat in Africa have been transformed by human activities, and the reduction or removal of populations of wild ungulates has contributed to the wild dogs' decline. Traffic and snares are responsible for significant additional sources of human related mortality in some parts of Africa (Woodroffe et al. 1997b).

Competing carnivores contribute to the enormous area requirements of wild dogs, and indirectly contribute to their conservation predicament. Lions are a significant source of mortality for wild dogs (up to 50% of adult deaths, Ginsberg et al. 1995) and limit access to habitats with high prey densities (Mills & Gorman 1997; Creel & Creel 2002). By virtue of their high metabolic requirements, wild dogs are highly susceptible to kleptoparasitism (Gorman et al. 1998), and in open habitats, spotted hyaenas limit wild dog populations through interference competition (Woodroffe et al. 1997b). Increasing populations of hyaenas and lions are believed to have been responsible for the decline in wild dog numbers in the Serengeti (Creel & Creel 1996), and the local extinction of wild dogs in Ngorongoro crater was probably due to direct competition with other carnivores (Creel & Creel 2002). Rabies and canine distemper are believed to have caused the final extinction of wild dogs in the Serengeti (Gascoyne et al. 1993), and disease is reported to have a limiting effect upon some populations of wild dogs. In South Africa, a rabies outbreak was responsible for the failure of the first attempt to reintroduce wild dogs into Madikwe Game Reserve (Hofmeyr et al. 2000). Across most populations of wild dogs, however, disease is not a major cause of mortality (van Heerden et al. 1995; Creel & Creel 1996).

1.3.2 Current conservation efforts

Although wild dogs are listed as 'Endangered' by the IUCN, the degree of protection afforded to the species varies greatly between range states, from total to non-existent (Woodroffe et al. 1997c). Within South Africa, the legal status of wild dogs varies between provinces, and in all areas the shooting of 'problem animals' in the defence of livestock is permitted. Across Africa, the most important strategy for the conservation of wild dogs in the long term is the maintenance of large protected areas (Woodroffe et al. 1997c), and the creation of 'trans-frontier parks' through the linking of neighbouring protected areas across national boundaries has significant potential for improving conservation status.

In South Africa, high human population densities, widespread habitat transformation and a shortage of suitable large protected areas limit scope for improving the conservation status of wild dogs. Land tenure conditions in South Africa create unique problems for the conservation of large carnivores: private farmland constitutes 73.4% of the national land area, compared to 45% in Namibia, 6% in Botswana and 0% in Tanzania (Cumming 1991), and the expansion of commercial agriculture resulted in the rapid extirpation of wild dogs from ~ 98% of South Africa's land surface. Historically, wild dogs were distributed from the south coast, throughout the former Cape Province, to the northern borders (Skinner & Smithers 1990). At present, only one viable population of wild dogs exists in South Africa, in the Kruger National Park (henceforth referred to as "Kruger") in the extreme northeast of the country.

Current efforts to improve the conservation status of wild dogs in South Africa have focused upon the creation of a meta-population through the reintroduction of wild dogs into geographically isolated reserves, linked by management (Mills et al. 1998). This strategy has achieved some success, and to date, six sub-populations have been established. Due to the limited number of suitable state-owned protected areas, the expansion of the meta-population is likely to depend increasingly upon private nature reserves.

In recent years, there has been a widespread shift from traditional cattle ranching to game ranching, whereby wild ungulates are utilised consumptively through hunting, and non-consumptively through ecotourism (Falkena 2000; van der Waal & Dekker 2000). A result of this has been a dramatic increase in the number of wild ungulates on private land, and conditions for the conservation of large carnivores have improved accordingly. In many areas, neighbouring game ranchers have cooperated through the removal of interior fencing to create private collaborative nature reserves or conservancies. The effect of these changing land use conditions is to provide scope for the expansion of the South African wild dog distribution outside protected areas onto private land.

1.4 This study

1.4.1 Objective

An understanding of the ecological, sociological and economic determinants of conservation success with regard to wild dogs outside state protected areas in South Africa.

1.4.2 Rationale

Wild dogs are South Africa's most endangered carnivore, and are limited to a single viable population, comprising <2% of their former range. Urgent conservation measures are required to increase the number and geographic range of wild dogs, and ultimately to improve their conservation status. As a result of changing land use patterns, conditions for conserving wild dogs outside state-protected areas are improving. Land outside state protected areas can be utilised to improve the status of wild dogs in two ways: through reintroductions into private nature reserves, and through the conservation of naturally occurring packs *in situ* on private livestock / game ranchland. These two scenarios are referred to repeatedly throughout the thesis, and are defined in the following sentences. a)

Reintroduction into private nature reserves is assumed to occur where wild dogs were absent, and into reserves from which they are prevented from leaving after release by the presence of predator proof fencing. Within this scenario, there is a range of private nature reserve types, from 'ecotourism' reserves in which predation by wild dogs is likely to result in no cost to the reserve owner, to reserves in which land use involves some consumptive utilisation of wildlife and predation is perceived to result in a direct cost to the reserve owner. b) Private nature reserves may occur within game / livestock ranchland, but the conservation of wild dogs *in situ* on ranchland is defined as occurring where wild dogs have re-colonised the area naturally, and where packs are largely able to pass between neighbouring ranches due to the absence of predator-proof fencing. Prior to the initiation of conservation efforts involving wild dogs outside state protected areas within either of the two scenarios, several feasibility aspects must be considered.

My study addressed two ecological issues relevant to the conservation of wild dogs outside state protected areas: First, I assessed the current status of wild dogs outside state protected areas, to determine what is left to conserve, and to identify the conditions in which wild dogs are persisting. Second, I determined the minimum ecological requirements for a pack of wild dogs to identify the minimum reserve sizes required for reintroductions, and the minimum habitat patch sizes required for *in situ* conservation efforts outside protected areas, assuming that genetic aspects are incorporated into conservation management, and that a pack is the demographic unit for wild dogs.

The success of conservation efforts outside protected areas is entirely dependent upon the cooperation of landowners, and their willingness to tolerate wild dogs on their land. I assessed the attitudes of landowners towards wild dogs, and identified the land tenure conditions under which conservation efforts are most likely to succeed.

The basis for the eradication of wild dogs from much of their former range is likely to have been real and perceived economic costs associated with livestock losses (Woodroffe & Ginsberg 1999). Economics form the basis for a substantial proportion of human behaviour (Shogrun et al. 1999), and consequently, an understanding of the economics associated with the conservation of wild dogs is a vital prerequisite for the design and implementation of conservation programmes. I assessed the economic costs associated with conserving wild dogs, and determined the extent to which costs can be offset by ecotourism related benefits. In this way, I identified the conditions under which wild dogs might effectively pay for their own conservation, and the conditions under which conservation efforts are likely to depend upon donor subsidies. In addition, I quantified the present subsidy to wild dog conservation from donor agencies, and utilised a cost efficiency approach to help design future conservation strategies.

1.4.3 Key questions

Specifically, my study aimed to answer the following key questions:

a) What is the present distribution and population status of wild dogs outside state-protected areas in South Africa?

b) What are the minimum area and prey requirements for a pack in the areas in which wild dogs occur in South Africa?
c) What are the attitudes of landowners towards wild dogs, and the reasons for these attitudes in the areas in which wild dogs occur on private land in South Africa?
d) What are the costs and potential benefits associated with conserving wild dogs within a viable population, through reintroduction into a reserve, and *in situ*, on ranchland?
e) To what extent has donor funding subsidised wild dog conservation in South Africa in recent years?
f) What is the most cost efficient strategy for improving the status of wild dogs in South Africa?

1.5 Overview of the thesis

The order of the chapters reflects the order of the key questions, outlined above. The thesis begins with an assessment of the distribution and status of wild dogs outside state protected areas in South Africa. Wild dog sightings were collected from a variety of sources from a period of six years, from January 1996 to June 2002. Geographic Information Systems technology enabled the analysis of wild dog distribution relative to environmental variables such as land cover, human population density and distance from source populations of wild dogs, and the identification of focal areas with the greatest potential for range expansion.

In Chapter 3, an attempt is made to identify the minimum areas required to support predation by an average pack of wild dogs. This method provided minimum area requirement estimates for wild dogs, and identified lower bound reserve sizes required for wild dog reintroductions. The utility of this approach is to assist in the selection of reintroduction sites for the expansion of the meta-population.

In Chapter 4, the results of interviews concerning rancher's attitudes towards wild dogs are presented. More than two hundred ranchers were interviewed from three parts of South Africa, and three parts of Zimbabwe. Results were used to identify the conditions under which conservation efforts are most likely to succeed.

In Chapter 5, the costs of conserving a pack of wild dogs within three scenarios were estimated: a) within a viable population, 2) through reintroduction into a private nature reserve as part of the expansion of the meta-population, and 3) the conservation of a naturally occurring pack *in situ*, on ranchland. A contingent valuation approach was used to determine the willingness of tourists to pay to see wild dogs. Estimates of the potential annual revenue from wild dog-based ecotourism were compared with costs under each conservation scenario, and the findings used to guide conservation and funding priorities.

Finally, in Chapter 6, donor-funding expenditure over the last five years was gauged with a survey of wild dog stakeholders. The way in which donor funds have been utilised was assessed relative to three sectors of the South African wild dog population: the Kruger population; the meta-population; and wild dogs occurring on ranchland. The efficacy of current conservation efforts was assessed, and potential future strategies suggested. A cost efficiency approach was employed to help determine which future strategy is likely to represent the best 'value for money'.

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CHAPTER 2

The distribution and population status of wild dogs *Lycaon pictus* outside protected areas in South Africa

2.1 Introduction

A variety of anthropogenic factors have resulted in a reduction in the numbers and geographic range of many carnivore species across the world (Gese 2001). The design and implementation of recovery programmes for these species is partially dependent upon rapid and accurate assessment of their abundance and distribution. Many carnivore species are nocturnal, secretive, rare and wide-ranging, making such assessment problematic (Gese 2001), despite a proliferation of methods to estimate carnivore range and abundance (Wilson & Delahay 2001). Members of the diverse large-carnivore guild in Africa have been a focus for the development of a variety of such techniques (e.g. lions *Panthera leo*, Pennycuik & Rudnai 1970, Smuts et al 1977; spotted hyaenas *Crocuta crocuta*, Mills 1985, Mills et al. 2001; and cheetahs *Acinonyx jubatus*, Gros et al. 1996).

Wild dogs are endangered and their conservation status has received particular attention as a result. During the last 30 years there has been a dramatic decline in the number and distribution of wild dogs across Africa, and at present, it is believed that viable populations occur in only six of the 34 countries in which they once occurred (Fanshawe et al. 1991; Woodroffe & Ginsberg 1999). Efforts have been made to document conservation status and population trends on various scales: continental, national, and by individual protected areas. Fanshawe et al. (1991) conducted a pan-African survey of wild dog status and distribution, and then Fanshawe et al. (1997) updated this with an intensification of methodology. Several studies have surveyed the status of wild dogs on a national level: Zimbabwe (Childes 1988); Botswana (Bulger 1990); Namibia (Hines 1990); Kenya (Jennings 1992); Zambia (Buk 1994); Ethiopia (Malcolm 1995); and Senegal (Sillero-Zubiri 1995). In South Africa, accurate assessment of wild dog populations was pioneered through the collection of photographic records of each individual in the Kruger National Park (henceforth referred to as "Kruger") population (Maddock & Mills 1994). Fanshawe et al. (1997) and Friedmann et al. (2002) estimated the distribution and status of wild dogs on a national level in South Africa, based upon a survey of field workers, and expert opinion combined with museum data, respectively. To date, however, no in depth attempt has been made to assess the number of wild dogs occurring specifically outside state protected areas in South Africa.

In South Africa, wild dogs were historically recorded from the southern coast, throughout the interior of the country to the northern borders (Skinner & Smithers 1990). Increasing human populations and commercial agriculture resulted in eradication from most of the historic range, and at present wild dogs are limited to a single viable population in Kruger (Stuart et al. 1985; Fanshawe et al. 1997).

There are no suitable protected areas of sufficient size for a second viable population in South Africa, and recent conservation efforts have concentrated upon establishing a meta-population, consisting of a number of sub-populations within a network of small reserves

(Mills et al. 1998). By 2002, sub-populations had been established in five protected areas, with a combined area of ~ 2,750 km^2 and a population size of approximately 54 adults and sub adults in 10 packs.

A shortage of suitable, large protected areas in South Africa suggests that land outside protected areas may need to be utilised to expand the geographic distribution of wild dogs. Land-use trends on private land in northern South Africa have seen a dramatic shift to game ranching from cattle ranching, resulting in increases in the distribution and numbers of many prey species (van der Waal & Dekker 2000), and a corresponding increase in the potential for conserving wild dogs. Determining the numbers and distribution of wild dogs currently occurring outside state protected areas represents a vital pre-requisite for conservation planning involving private ranchland.

2.2 Methods

The large spatial scale of the study, restricted access over large areas, and secretive behaviour of the dogs outside state protected areas precluded implementation of the census techniques used in Kruger (Maddock & Mills 1994). The collection of sighting records is a method widely implemented to survey the status of large carnivores (Gese 2001) and was used to estimate the numbers and distribution of wild dogs outside state protected areas (excluding privately owned land incorporated into national parks). Sighting records recorded between January 1996 and August 2002 were included, from the following sources:

a) Representatives from provincial conservation authorities (hereafter referred to as nature conservation). Landowners frequently report the presence of large carnivores to nature conservation authorities, and detailed records of wild dog sightings were obtained from officers in charge of 'problem animal' control in each province (except for the Eastern and Western Cape, where sightings are extremely unlikely). Where possible, contact numbers of informants were obtained and follow up calls made to obtain detailed sighting-information.

b) Appeals for sightings published in the English and Afrikaans media, including: three national agricultural / wildlife magazines; two wildlife newsletters; and 14 local newspapers with a wide coverage (Appendix A). In each article, a photograph of a wild dog was included to help prevent misidentification.

c) Interviews with ranchers from three focal areas where published records mention recent wild dog activity outside protected areas: i) ranching areas along the Limpopo River (Skinner & Smithers 1990); ii) ranching areas west of Kruger (Fanshawe et al. 1997); and iii) ranching areas north of Hluhluwe-Umfolozi Park (Maddock 1999). Focal areas of activity were demarcated on a 1 / 250,000 map with the assistance of nature conservation representatives, yielding approximate central coordinates as follows; i) 22° 20' S, 29° 40' E; ii) 24° 10' S, 30° 55' E; iii) 27° 30' S, 31° 45' E, respectively. Ranch names were obtained from 1 / 250,000 maps, and corresponding telephone numbers derived from telephone directories. In each area, as many ranchers as possible were personally interviewed with a set questionnaire (Appendix B), in a two-week period. A total of 166 ranchers was interviewed in all three areas.

d) Interviews with local communities in communal land in which nature conservation authorities reported wild dog presence. The head ecologist (Chris Roche) of a large safari operator (CCAfrica) asked managers and staff working in safari camps within Timbavati Game Reserve on the western border of Kruger (Ngala Lodge), and Mtethomusha Game Reserve (Bongani Lodge), on the southwestern border of Kruger, to provide information on any sightings they heard of from the neighbouring communal lands where they live. This provided coverage of large areas of communal land neighbouring Kruger, with approximate central coordinates of 24° 40' S, 31° 10' E, and 25° 20' S, 30° 10' E.

e) Networking with field workers. Field researchers and wildlife capture teams working in the northern half of the country were contacted and asked to provide information on any sightings they heard of. Additional appeals for information were made at Wild Dog Advisory Group-South Africa meetings.

Respondents were asked to provide as many details as possible for each sighting, including: 1) date; 2) location; 3) land use at location of sighting; 4) number of individuals; 5) sexes; 6) ages (juvenile, sub adult, adult), and 7) frequency of sightings. When wild dogs were reported away from areas of high sighting frequency, respondents were asked for a description of the animal(s) seen to determine whether they had correctly identified the species sighted. If wild dogs were seen on a given property for less than one month, a single sighting was recorded irrespective of the actual number of sightings made. If wild dogs utilised a property over several months, a single sighting was recorded for each month that they were present. Sightings were digitized using ArcInfo (version 3.2).

2.2.1 Total number of wild dogs outside protected areas

The largest home range recorded in Kruger is 1,110 km^2 (Reich 1981) and this area was used as a basis for distinguishing between sightings of different groups. The diameter of 1,110 km^2 is 37.5 km, assuming the home range approximates a circle. Sightings were ordered by date and the distance between sequential sightings was measured. Sightings that fell within 37.5 km of the previous sighting were assumed to constitute the same group. A repeat of these methods based upon the mean Kruger home range size (537 km^2, Mills & Gorman 1997) yielded a very similar estimate of the number of groups, suggesting that within reasonable limits the method is insensitive to the home range area used.

2.2.2 Resident versus dispersing wild dogs – upper and lower limits

Summing the mean size of each group yielded an upper limit for the estimated number of wild dogs. Counting the mean pack size of wild dogs resident outside state protected areas (successful colonists), and excluding dispersing groups (potential colonists) yielded a lower limit of the estimated number of dogs. Wild dogs of 1 - 2 years old disperse in single sex groups, up to 250 km from natal home ranges (Fuller et al. 1992a). Most respondents, although able to distinguish between adults and pups, were unable to distinguish between adults and sub adults, and most failed to provide details of the sexes of dogs seen. Consequently, group size was used as the basis for distinguishing between resident and dispersing dogs. The average recorded size of dispersing groups is 2.3 – 3.4 individuals for females, and 3.9 – 5.3 individuals for males (McNutt 1996; Creel & Creel 2002). Wild dogs were assumed to comprise dispersing groups if sightings of ≤ 5 individuals were received, or resident packs if sightings of ≥ 6 individuals were received, or if the presence of puppies or den sites were reported. An exception was made if a group of <6 wild dogs was sighted within a home range area (as defined above) in which a 'resident pack' was seen in the previous year. Mean group size was calculated by averaging the modal group size reported for each pack. Variation in the number of sightings of each pack prevented the use of an overall mean, and necessitated the calculation of the modal number of dogs in each pack, which was then averaged. Wild dogs sighted within 20 km of the Kruger boundary were arbitrarily assumed to be part of the Kruger population, irrespective of group size, unless >3 sightings were reported.

2.2.3 Geographic distribution

ArcInfo was used to create two measures of geographic distribution, extent of occurrence, and area of occupancy (IUCN 2001). Extent of occurrence was calculated using a minimum convex polygon comprising the area contained within the shortest continuous boundary encompassing all sightings of resident wild dogs (Appendix C). Area of occupancy was calculated by drawing a polygon around the outer most sightings of resident wild dogs (Appendix D). A pack of resident dogs sighted only once, in the North West province, was assumed to occupy an area equal to the largest home area observed in

Kruger (1,110 km^2, Mills & Gorman 1997) and this figure was added to both the extent of occurrence and area of occupancy estimates. To prevent exaggerated estimates of geographic distribution, sightings of dispersing wild dogs were excluded.

2.2.4 Wild dog distribution relative to environmental variables

A map of South Africa, north of the most southerly recorded sighting (approx. 30° 30' S) was converted to a grid cell format, (5 km^2 cells), and each cell denoted a code: 0 for no sightings ('absent cells'); 1 for dispersing wild dogs sighted ('dispersing cells'); and 2 for resident wild dogs sighted ('resident cells'). The relationship between wild dog distribution, land cover (degree of habitat modification) and human density was investigated by superimposing the grid cell map upon the CSIR National Land Cover Database (Thompson 1999), and 2002 human population census data (South African Municipal Demarcation Board 2002). When analysing wild dog occurrence relative to land cover, it was assumed that all habitat unmodified by human activity represented potentially suitable habitat. The original 31 land cover categories in the CSIR land cover database were categorised as; 'suitable', 'degraded', or 'unsuitable' based upon the degree of transformation (Appendix E).

The relationship between wild dog distribution and the distance from source populations was investigated by determining the shortest distance from the centre of each grid square to the nearest boundary of one of three potential source populations (Kruger, Hluhluwe-Umfolozi Park and Central Kalahari Game Reserve in Botswana).

For each grid cell, these methods provided an estimate of wild dog status (absent, dispersing, resident), land cover, human density, and distance from a source population. The relationship between these variables and wild dog status was investigated using ordinal logistical regression. Given the wide habitat tolerance of wild dogs (Skinner & Smithers 1990), natural vegetation type was not included as a variable in this analysis.

Data describing the distribution of land use on a fine scale (e.g. cattle ranching versus game ranching) on a national level was unavailable, and this factor could not be analysed with the other variables. However, the land use at each sighting was recorded, and the relationship between sightings of resident versus dispersing wild dogs and land use was analysed using ordinal logistical regression. For the analysis, land use at the location of each sighting was categorised as: communal land; livestock ranching (cattle or sheep); mixed cattle / game ranching; or game ranching (including private nature reserves). A note was made for each sighting as to whether the location of the sighting was part of a collaborative nature reserve, where neighbouring ranchers had cooperated to remove internal fencing.

2.2.5 Available habitat for expansion of the wild dog population outside protected areas

The area of remaining unmodified natural habitat, with a human population density of ≤ 5 people / km^2 (the modal population density of cells with sightings of resident dogs) north of the most southerly sighting was calculated using ArcInfo to provide an estimate of the area of suitable habitat for wild dog conservation in northern South Africa (excluding private land which has been incorporated in to the greater Kruger, state reserves, and private nature reserves into which wild dogs have been reintroduced).

2.3 Results

A total of 516 sightings were reported from provincial nature conservation representatives (38.3% of sightings), field workers (28.6%), interviews with ranchers (23.2%), responses to publications (5.6%) and community interviews (4.3%).

2.3.1 Total number of wild dogs outside state protected areas

The upper limit of the wild dog population estimate was 84.9 ± 7.97 (mean ± S.E.) individuals between 1996 - 2002, varying from a low of 42 in 1996 to a high of 106 in 2000 (Table 2.1). The lower limit averaged 54.7 ± 5.54 individuals between 1996 - 2002, and varied from a minimum of 25 in 1996 to a maximum of 67 in 1997. The large estimated number in 1997 relative to 1996 was due in part to the appearance of a large pack in northwestern Limpopo that year. The number of wild dog groups (resident packs plus dispersing groups) varied from a minimum of 10 in 1996 and 1997 to a maximum of 21 in 2000 (Table 2.2). The estimated number of resident packs ranged from a minimum of four in 1996 to a maximum of 10 in 2000 and 2001. Twenty-nine den sites were reported between 1996 and 2002.

Table 2.1 The number of resident and dispersing wild dogs occurring outside protected areas during 1996 - 2002 (number of sightings in parentheses)

	1996 (47)		1997 (31)		1998 (40)		1999 (55)		2000 (93)		2001 (160)		2002 (90)	
	[a] Res	[b] All	Res	All	Res	All	Res	All	Res	All	Res	All	Res	All
Kwa-Zulu Natal	5	7	5	5	8	8	5	6	8	9	3	4	0	1
Mpumalanga														
Kruger border	0	6	0	1	5	11	5	16	4	16	0	9	0	0
Highveld	2	2	7	7	7	7	7	7	5	5	4	4	6	10
Limpopo														
Kruger border	11	18	21	34	25	32	12	22	11	22	14	22	12	12
Limpopo River	7	7	12	12	4	4	26	26	31	33	23	29	15	15
Northwest Limpopo	0	0	22	27	10	18	0	12	1	14	10	14	9	13
North West	0	2	0	0	0	7	0	5	0	7	10	16	6	22
Northern Cape	0	0	0	5	0	0	0	0	0	0	0	0	0	3
TOTAL	25	42	67	90	59	87	55	95	65	106	64	98	48	76

[a] Resident.
[b] Resident and dispersing.

Table 2.2 The number of resident packs and dispersing groups occurring outside protected areas during 1996 - 2002 (number of sightings in parentheses)

	1996 (47)		1997 (31)		1998 (40)		1999 (55)		2000 (93)		2001 (160)		2002 (90)	
	[a] **Res**	[b] All	**Res**	All	**Res**	All	**Res**	All	**Res**	All	**Res**	All	**Res**	All
Kwa-Zulu Natal	1	2	1	1	1	1	1	2	1	2	1	2	0	1
Mpumalanga														
Kruger border	0	2	0	0	1	3	1	3	1	3	0	2	0	0
Highveld	1	1	1	1	1	1	1	1	1	1	1	1	1	2
Limpopo														
Kruger border	1	2	2	3	2	3	2	3	2	3	2	3	2	3
Limpopo River	1	1	2	2	1	1	2	2	4	5	3	5	2	2
Northwest Limpopo	0	0	1	2	1	3	1	3	1	5	2	3	1	3
North West	0	2	0	0	0	2	0	1	0	2	1	4	1	5
Northern Cape	0	0	0	1	0	0	0	0	0	0	0	0	0	1
TOTAL	4	10	6	10	7	14	8	15	10	21	10	20	7	17

[a] Resident.

[b] Resident and dispersing.

2.3.2 Group sizes

Mean group size (± S.E.) across all sightings was 6.9 ± 0.64 dogs and was very similar to the average pack size recorded in Kruger in the latest census (7.1 dogs, Davies 2000). The mean size of groups containing only adults and sub adults was 5.0 ± 0.34 S.E. dogs.

2.3.3 Wild dog distribution

Sightings of resident wild dogs during 1996 and 2002 had an area of occupancy of 17,907 km^2, and an extent of occurrence of 43,310 km^2. Most sightings (Figure 2.1) came from the ranching area on the western border of Kruger in Limpopo (n = 244). Wild dogs persisted in the area despite frequent persecution - five and seven individuals were reported shot in 1997 and 1998 respectively, and a pack was captured and removed in 1999. During 1996 - 2000 one to two packs were resident in the area, comprising 11 to 25 individuals (Tables 2.1, 2.2), with an area of occupancy of 4,247 km^2, and an extent of occurrence of 8,850 km^2. The majority of sightings (Table 2.3) occurred on game ranches (89.8% of sightings), with few sightings in communal land (8.6%). Almost 50% of sightings (48.7% of sightings) were made on collaborative nature reserves. One pack normally resident in Kruger had a den site in communal land neighbouring Kruger during the winter months of 1996 - 2001.

Sightings were also frequently reported from along the Limpopo River, close to the border with Zimbabwe (n = 136). The source of these dogs was probably northern

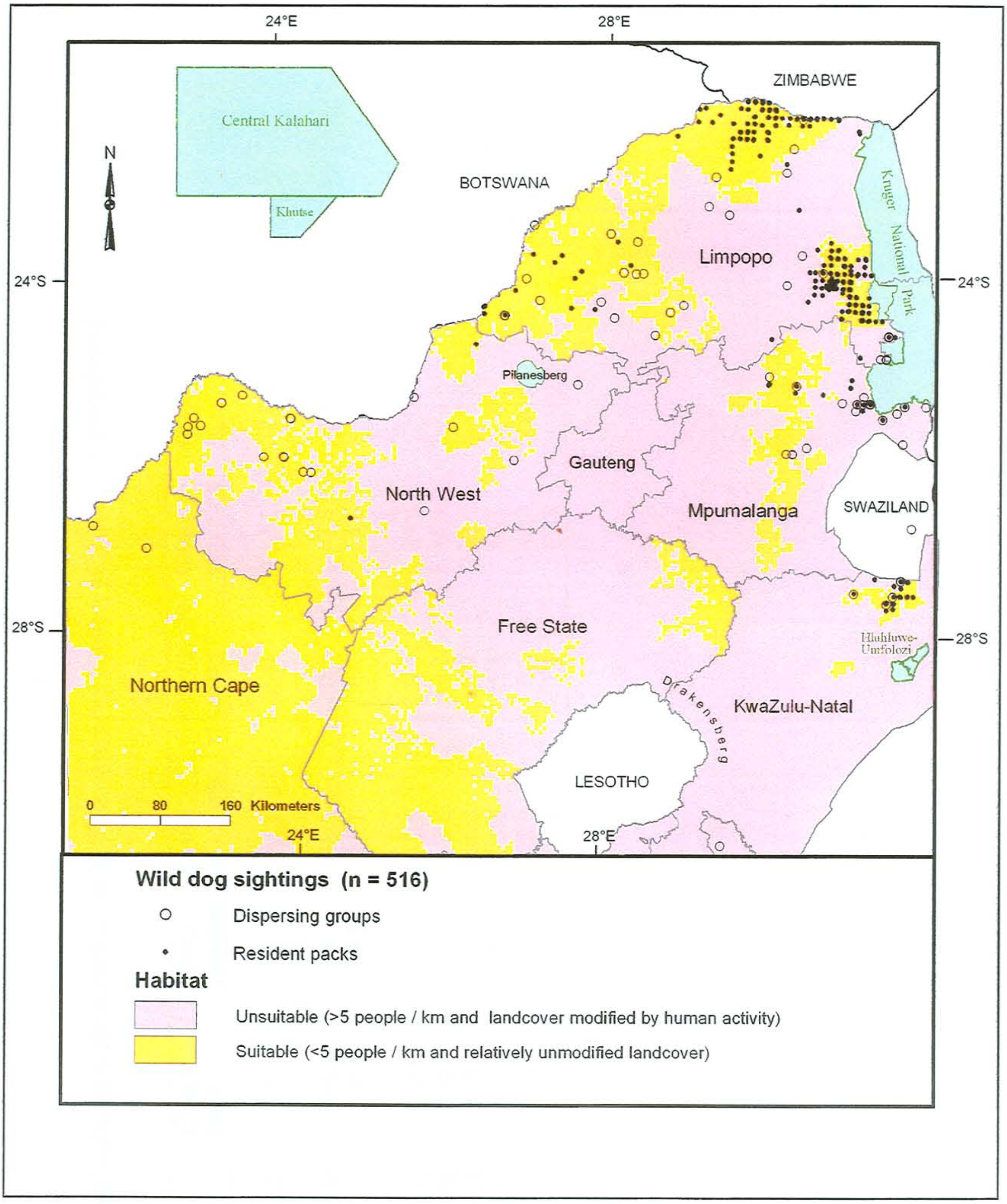

Figure 2.1 Relationship between the distribution of wild dogs outside protected areas and suitable habitat

Table 2.3 The percentage of sightings made in each of six land use categories (number of sightings in parentheses)

		Communal land	Sheep farming	Cattle ranching	Cattle / game ranching	Game ranching	Other
Kwa-Zulu Natal	(32)	0	0	3.1	25.0	53.1	18.8
Mpumalanga							
Kruger border	(28)	34.8	0	13.1	0	47.8	4.3
Highveld	(18)	0	83.3	11.1	0	5.6	0
Limpopo							
Kruger border	(244)	8.6	0	1.6	0	89.8	0
Limpopo River	(136)	0	0	3.8	9.7	84.9	1.6
Northwest Limpopo	(35)	0	0	58.8	5.9	32.4	2.9
North West	(18)	0	15.4	23.1	15.3	46.2	0
Others	(5)	20.0	0	20.0	0	40.0	20.0
OVERALL	(516)	6.0	4.0	5.8	7.5	69.5	7.2

Kruger, or the nearby Gona-re-zhou National Park in Zimbabwe. During 1996 - 2002 one to four resident packs were sighted in this area, comprising 4 - 31 individuals (Tables 2.1, 2.2), with an area of occupancy of 4,900 km^2 and an extent of occurrence of 11,525 km^2. In addition, one to two dispersing groups were infrequently sighted each year. Most sightings (Table 2.3) occurred on game ranches (84.9% of sightings), followed by mixed cattle / game ranches (9.74%) and cattle ranches (3.8%). A quarter of sightings (25.1% of sightings) were made on collaborative nature reserves. Several attempts have been made by nature conservation authorities to remove wild dogs from the area over the last five years. Two sets of puppies were removed from dens in 1997, in 1999 four adults were captured and removed, and in 2001, five adults were removed. In addition, part or all of a pack of was illegally captured and sold by ranchers in 2001. Although no proven cases of persecution were reported, ~ 4% of ranchers interviewed indicated without prompting that they would shoot wild dogs on their properties (Chapter 4).

Sightings were recorded in Mpumalanga (46 sightings), with resident dogs occurring in an area of occupancy of 2,650 km^2, with an extent of occurrence of 7,775 km^2. There was an aggregation of sightings along the southern and southwestern border of Kruger (28 sightings), most of which were probably groups dispersing from the park, or packs ranging beyond the park periphery. However, one resident pack occurred in this area between 1998 and 2000 (Table 2.2). Most sightings (Table 2.3) occurred on game ranches (47.8% of sightings), and communal land (34.8%). Thirty four percent of sightings were made on collaborative nature reserves. On two occasions (1998, 1999) small groups were captured outside Kruger in Mpumalanga and returned to the park.

Eighteen sightings were reported from the Mpumalanga Highveld, approximately 80 km west of the park. A single resident pack occurred in the area between 1996 and 2002 and was mostly seen on sheep farms (83.3% of sightings) and cattle ranches (11.1%, Table 2.3).

Thirty-two sightings were reported from northern Kwa-Zulu Natal, of wild dogs likely originating from Hluhluwe-Umfolozi Park (Fanshawe et al. 1997), which lies ~ 75 km to the south. In this area, most sightings occurred on game ranches (53.1% of sightings, Table 2.3), all of which belonged to collaborative nature reserves, and mixed cattle / game ranches (25.0%). Sightings consisted of one resident pack between 1996 - 2001, with an area of occupancy of 975 km^2 and an extent of occurrence of 1,825 km^2, as well as dispersing groups or individuals in 1996, 1999, 2000 and 2001. A singe male was sighted in 2002, which has since died. Further sightings have not been reported, and wild dogs are probably locally extinct.

Additional sightings were reported from northwestern Limpopo, North West and the Northern Cape. Sightings in these areas were infrequent, widely dispersed, and consisted primarily of dispersing groups. One resident pack was reported from northwestern Limpopo in 1997, 1998, 1999, 2000 and 2002, with two resident packs being reported in 2001. Most of these sightings (Table 2.3) occurred on cattle ranches (58.8% of sightings) and game ranches (32.4%). Only 7.8% of sightings occurred on collaborative nature reserves. In North West, a resident pack was sighted in 2001 and 2002. Although most sightings occurred on game ranches (46.2% of sightings), several sightings occurred on cattle ranches (23.1%) and mixed cattle / game ranches (15.3%). No sightings occurred on collaborative nature reserves. A pack of five wild dogs sighted in the Northern Cape in 1997 was shot soon after by a rancher in the same province, while two individuals from a group of four were reported shot in North West in 2002.

2.3.4 Relationship between wild dog sightings and environmental variables

Using ordinal logistical regression, a relationship was found between land cover, human density, distance from a source population, and the presence of resident versus absent cells ($\chi^2 = 578.34$, df = 8, p<0.0001), the presence of dispersing versus absent cells ($\chi^2 = 73.17$, df = 7, p<0.0001), and the presence of resident versus dispersing cells ($\chi^2 = 76.82$, df = 5, p<0.0001, Table 2.4). A greater proportion of resident cells occurred in suitable land cover than absent cells (p = 0.020), or dispersing cells (p = 0.038). A greater proportion of dispersing cells occurred in suitable land cover than absent cells (p = 0.0045). Over 91% of resident cells occurred in suitable land cover, with 4.7% in degraded, and 4.1% in unsuitable land cover. Of dispersing cells, 80.4% occurred in suitable land cover, 5.3% in degraded, and 14.3% in unsuitable land cover. For absent cells, these figures were 74.8%, 5.8% and 19.4% respectively.

Human density was lower in resident than absent cells (p<0.0001), or dispersing cells (p = 0.604), and lower in absent than dispersing cells (p = 0.760). In resident cells, mean

Table 2.4 The relationship between the occurrence of wild dogs, human density, land cover and distance from source populations

	Absent cells	Dispersing cells	Resident cells
Percentage land cover			
Suitable	74.8	80.4	91.2
Degraded	5.8	5.3	4.7
Unsuitable	19.4	14.3	4.1
Human density (people / km^2)			
Mean ± S.E.	47.1 ± 2.2	77.5 ± 44.5	19.5 ± 4.8
Maximum	2390	2428	700
Distance from a source population (km)			
Mean ± S.E.	340 ± 1.2	187 ± 16.1	86.6 ± 5.0
Maximum	795	416	298

human density was 19.5 ± 4.77 (S.E.) people / km^2, compared to 47.1 ± 2.24 people / km^2 in absent cells and 77.5 ± 44.5 people / km^2 in dispersing cells (Table 2.4). There was an interaction between human density and land cover in the 'resident versus absent' model ($p<0.0001$), by virtue of the fact that 10.1% of resident cells occurred in unsuitable habitat, in which mean human density was 73.8 ± 40.1 people / km^2, compared to 13.3 ± 2.31 people / km^2 for the 89.9% of resident cells occurring in suitable habitat. There was an interaction between human density and distance from a source population in the 'resident versus absent' ($p = 0.0005$), and 'dispersing versus absent' ($p = 0.0030$) models as a result of higher human densities in cells closer to source populations.

Resident cells were generally closer to source populations than dispersing ($p = 0.001$) or absent cells ($p<0.0001$), and dispersing cells were typically closer than absent cells ($p<0.0001$). The mean distance of resident cells from a source population was 86.6 km^2 ± 5.02 (S.E.), compared to 187 km^2 ± 16.1 for dispersing cells, and 340 km^2 ± 1.17 for absent cells.

A relationship also existed between the presence of resident versus absent wild dogs, and land use ($\chi^2 = 63.7$, df = 2, $p<0.0001$). The large majority of sightings of resident wild dogs occurred on game ranches (77.2% of sightings, Figure 2.2), whereas sightings of dispersing wild dogs were spread more evenly across game ranches (32.9%), livestock ranches (sheep and cattle, 26.3%), and communal land (20.0%).

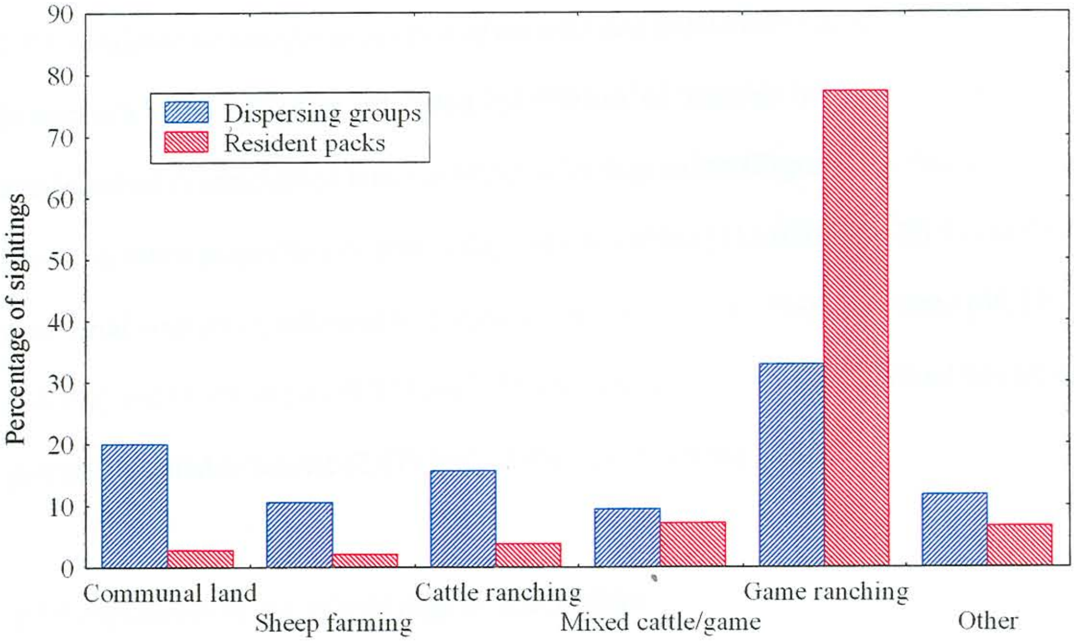

Figure 2.2 The percentage of sightings of resident and dispersing groups of wild dogs made in each of six land use categories

2.3.5 Available habitat for expansion of the wild dog population outside protected areas

In northern South Africa, an estimated 264,900 km^2 of 'suitable habitat' (<5 people / km^2 and (relatively) unmodified land cover) for wild dogs exists (Figure 2.1). Northern Cape has the greatest proportion of potentially suitable habitat (115,650 km^2 - 90.5% of the provincial land area), followed by Limpopo (46,550 km^2 - 46.3%), Free State (49,350 km^2 - 38.5%) and North West (39,725 km^2 - 34.4%). In contrast, Kwa-Zulu Natal has little potentially suitable habitat (2,475 km^2 - 3.4%), and Gauteng none.

2.3.6 Population status of wild dogs in South Africa

Latest census figures indicate that 177 wild dogs occur in Kruger in 21 packs (Davies 2000), down from 434 in 32 packs 1995 (Wilkinson 1995) probably due to a natural fluctuation. Prior to the denning season in 2002, an additional 54 wild dogs in 10 packs occurred in the meta-population, spread over five reserves (Hluhluwe-Umfolozi Park, 15 individuals; Karongwe Game Reserve, four; Madikwe Game Reserve, 16; Pilanesberg National Park, 11; Venetia Limpopo Nature Reserve, eight). Based upon these estimates, in 2002 279 - 307 free ranging wild dogs occurred in 37 - 47 packs in South Africa, with 57.7% of individuals occurring in Kruger (44.6% of packs), 24.7% outside protected areas (34.0% of packs), and 17.6% in the meta-population (21.3% of packs).

2.4 Discussion

The collection of sighting data seems to be a useful technique for estimating the distribution and status of wild dogs outside state protected areas in South Africa. Wild dogs are well suited to the collection of sighting data for census purposes, as they are gregarious, diurnal, and charismatic enough to make sightings memorable (Gros 1998). It was noted in the latest census of the Kruger population, that the average pack size calculated from tourist sighting reports was very similar to that calculated from photographic evidence (differing by 0.67 dogs, Davies 2000), suggesting that sighting records yield accurate group size estimates. However, it is important to note the limitations of this method. There is no way of measuring the associated error, and the assumptions made may affect the validity of the results. Population size estimates were based upon the assumption that all sightings within a distance equal to the diameter of maximum Kruger home range area from the previous sighting constituted a single pack. It is possible that the presence of fences between ranches limits movement, and that the high stocking rates typical of game ranches results in small home ranges. If this is the case, the number of packs was underestimated. However, several factors suggest that the use of large home range areas as a basis for distinguishing between packs is likely to be valid. First, high quality electric fencing is required to hold wild dogs, and movements are unlikely to be significantly affected by the fencing typically found on cattle and game ranches (Hofmeyr 2000). Second, home ranges are largest where there are few other wild dogs to impede movement (Fuller et al. 1992b) and consequently home ranges outside state protected areas are likely to be large. Third, the degree of overlap between home ranges of neighbouring packs is likely to be less given discontinuous natural habitat. Finally, cheetahs utilise larger areas in areas where they are persecuted (Marker et al. in prep.) and the same is likely to be true for wild dogs.

It is possible that the use of group size to distinguish between resident packs and dispersing groups underestimated the number of resident packs, as the mean size of female dispersing groups (2.3 - 3.4, McNutt 1996; Creel & Creel 2002) is smaller than the cut off figure of five used to distinguish between dispersing groups and resident packs. However, sightings of resident packs were reported 8.2 ± 0.48 (mean $\pm$ S.E.) times on average, compared to 1.6 ± 0.14 times for dispersing groups, suggesting that dispersing groups were correctly identified as being non resident.

Distribution and population size projections are minimum estimates, and blank areas on the distribution map do not necessarily mean that no wild dogs occur, simply that no sighting reports were received. For example, it is likely that wild dogs occur more frequently than reported in the Madimbo corridor on the extreme northwestern boundary of Kruger. This area is managed by the South African Defence Force, and due to the low density of people, sightings are less likely to be reported than from other areas. However, wild dogs are highly visible, and although dispersing groups undoubtedly pass through blank areas on the distribution map, resident packs probably do not occur far beyond the limits of the distribution presented. When wild dogs were sighted in an area, their presence was often corroborated by sighting reports from multiple sources. Furthermore, by virtue of their being highly visible, wild dog sighting data may be less affected by biases related to varying sightability between habitat types than studies of more secretive species such as cheetah (Gros et al. 1996) or cougar *Felis concolor* (Pike et al. 1999). Nonetheless, although distribution and population size estimates are probably conservative, the validity of these methods has not been tested for wild dogs, and the results obtained must be treated with caution.

The geographic distribution of wild dogs outside state protected areas in South Africa is greater than that suggested by Fanshawe et al. (1997) and Friedmann et al. (2002). Friedmann et al. (2002) estimated an extent of occurrence of 20,000 km^2, and an area of occupancy of >2,001 km^2, compared to my estimates of 43,310 km^2 and 17,907 km^2, respectively. The presence of vagrants in the Northern Cape and resident wild dogs on ranchland north of Hluhluwe-Umfolozi Park is confirmed (Fanshawe et al 1997; Maddock 1999), but the distribution and numbers occurring on ranchland along the Limpopo River and on the western border of Kruger is greater than previously suggested (Fanshawe et al. 1997). Sighting reports were received from several areas not mentioned by Fanshawe et al. (1997), including the southern and southwestern periphery of Kruger, Mpumalanga Highveld, northwestern Limpopo and North West.

Nonetheless, despite relatively widespread sightings of dispersing groups, the occurrence of resident wild dogs is limited primarily to areas close to source populations, with low human densities and unmodified habitat, dominated by game ranching. The three areas in which resident wild dogs are most commonly sighted (along the Limpopo River, the western border of Kruger, and northern Kwa-Zulu Natal) are all within 90 km of a source population. Kruger is a much larger source population than Hluhluwe-Umfolozi, and this is borne out by the greater number of sightings around Kruger. Dispersal greatly increases mortality risk in wild dogs (Creel & Creel 2002), and survival rates are likely to decrease with distance from a source population due to exposure to unsuitable blocks of habitat, tarred roads, disease-risk through contact with domestic dogs *Canis familiaris*, and persecution. High human densities surround much of northern and southern Kruger and Hluhluwe-Umfolozi Park and provide an immediate barrier to dispersal. Correspondingly, dispersing cells had higher average human densities than absent cells, and had varied land cover (several sightings even occurred within town limits) and varied land use, due to the necessity of crossing inhospitable terrain to reach suitable habitat. This coupled with an Allee effect operating at low pack sizes (Courchamp & Macdonald 2001), and a paucity of source populations may explain why large areas of apparently suitable habitat (such as northwestern Limpopo) have not been successfully re-colonised. Wolves *Canis lupus* in North America, by contrast, have been observed to successfully disperse up to 600 km over agricultural land (Mech 1995). Given landowner cooperation, the translocation of wild dogs to areas of suitable habitat would greatly assist range expansion.

Wild dogs resident outside protected areas occur primarily on private land used for game ranching. Under these conditions, natural vegetation is typically intact and prey populations are actively protected. In contrast, few sightings were reported on communally owned property, likely as a result of depleted prey populations (Bigalke 2000). In some areas, notably the ranching area on the western border of Kruger, and in Kwa-Zulu Natal, a large portion of sightings (48.7% of sightings, and 53.1% respectively) was made on collaborative nature reserves. Most ranchers within collaborative nature reserves are positive towards wild dogs (Chapter 4) and this form of land use provides conditions highly conducive to their conservation. An estimated 13% of South Africa is devoted to game ranching (Falkena 2000), suggesting that significant potential for wild dog conservation exists outside protected areas. However, this potential is reduced by habitat fragmentation - few large blocks of unmodified habitat exist outside protected areas. Ranging behaviour predisposes wild dogs to edge effects and increased mortality risk (Woodroffe & Ginsberg 1998) and approximately 9% of sightings of resident wild dogs occurred in conditions conducive to conflict with people: unsuitable habitat with high human densities and livestock based land uses.

Wild dogs outside state protected areas are limited largely to areas with low human population densities, in keeping with other carnivore species (e.g. cougars, Pike et al. 1999). By virtue of their cooperative breeding strategy, wild dogs are highly sensitive to adult mortality, and intolerant of persecution (Courchamp & Macdonald 2001). However, in some areas, resident packs persist despite high levels of human-related mortality, suggesting that wild dogs may be more tolerant of persecution than previously believed. Furthermore, resident packs persist at higher human densities (mode, 5.0 people / km^2; mean, 20 people / km^2; range, 1.5 - 150 people / km^2) than the critical density, above which extinction is predicted (0.7 people / km^2), and within the range of mean densities at which other large carnivores have gone extinct (e.g. wolves, 13.5 people / km^2; lions, 26.0 people / km^2, Woodroffe 2000). Nonetheless, persecution by humans probably limits the expansion of the distribution of wild dogs outside state protected areas, and reducing human-related mortality must be a focus of conservation efforts. Recent recoveries in large carnivore populations in Europe and North America suggest that successful conservation is possible at high human densities given adequate law enforcement (Linnell et al. 2001).

2.4.1 Potential for range expansion

The best prospects for range expansion outside protected areas are probably in northern and northwestern Limpopo, and in northeastern North West province, where large areas of contiguous suitable habitat persist, with low human densities, largely unmodified land cover, and large prey populations due to a prevalence of game ranching (van der Waal & Dekker 2000). Significantly, cheetahs, lions, leopards *Panthera pardus*, and spotted hyaenas occur in these areas, suggesting that the prevailing conditions are conducive to large carnivore conservation (Friedmann et al. 2002). Given landowner tolerance, increasing wild ungulate populations in northern South Africa have the potential to support increases in wild dog numbers in these areas. However, ranchers in northern Limpopo are largely negative towards wild dogs, and in the absence of education programmes, and schemes aimed at reducing the costs and increasing the economic benefits associated with conserving the species, range expansion is unlikely to occur (Chapter 4).

There is some potential for range expansion in the Eastern Cape through translocation of wild dogs into Addo Elephant Park, or one or several of the growing number of private nature reserves in the region. Large areas of unmodified habitat with low human densities also occur in the Northern Cape. However, wild dogs are only vagrant in the Kgalagadi Transfrontier Park and the habitat in the remainder of the Northern Cape is probably marginal or unsuitable for wild dogs (Fanshawe et al. 1997). In the western North West province and Northern Cape, wild dogs were often sighted on livestock ranches, where one would expect prey densities to be low. Reduced prey availability is associated with increased transience in carnivore populations (Fuller & Sievert 2001), and a shortage of suitable prey may explain the lack of resident packs in these areas.

If one conservatively assumes that all habitat in the Northern Cape and Free State provinces is unsuitable for wild dogs, and that suitable habitat is equal to the area of unmodified land with fewer than 5 people / km^2 in Kwa-Zulu Natal, Limpopo, and North West, then an area of 88,750 km^2 is potentially available for wild dog conservation outside state protected areas. The population size of wild dogs that could be conserved in this area is 178 individuals (or ~ 18 packs) given a density equal to the lowest density recorded in a protected area (two dogs / 1000 km^2, Fuller et al. 1992b), or 1,482 individuals (or ~ 148 packs) given a density equal to the minimum density observed in Kruger (16.7 dogs / 1000 km^2, Maddock & Mills 1994). Although in reality, some of habitat classified as 'suitable' in this study is likely to be unsuitable due to low prey populations and incompatible land uses, there is nonetheless significant scope for expansion of the distribution of wild dogs outside state protected areas.

2.4.2 Total South African wild dog population

Despite being frequently cited as one of six countries with a large and stable wild dog population (Fanshawe et al. 1997; Creel & Creel 2002), numbers in South Africa are precariously low, with a geographic distribution covering as little as 3.54% - 5.69% of the country, based upon the area of occupancy and extent of occurrence, respectively. The 2002 population estimate of 279 - 307 wild dogs is markedly lower than that for other large carnivores in South Africa; cheetah (650 individuals), lions (2,519 individuals) and spotted hyaenas (2,970 individuals, Friedmann et al. 2002). The recent decline in wild dog numbers in Kruger probably represents a natural fluctuation likely related to high rainfall and low hunting success (Davies 2000), and numbers are believed to be recovering (G. Mills pers. comm.). Nonetheless, a population size of 177 leaves a demographically effective population size of as low as 140 (due to unbalanced sex ratios, and deviation from a stable age distribution), and the Kruger population is susceptible to environmental and demographic stochasticity (Caughley 1994; Creel & Creel 2002). The attempt to create a second viable population through the establishment of the meta-population had yielded an additional 54 individuals in 10 packs by 2002. Wild dogs outside state protected areas form a more significant sector of the national population than previously realised (Fanshawe et al. 1997; Friedmann et al. 2002). Conservationists and donors have neglected wild dogs occurring outside state protected areas, and they are highly persecuted, remain low in numbers and are limited in distribution. This neglect likely affects populations within protected areas by creating a 'vacuum effect' (Rasmussen 1999) whereby wild dogs leave the safety of protected areas to fill empty home ranges vacated by persecution. The channelling of donor funding and conservation efforts into wild dog conservation outside protected areas is vital to facilitate range expansion, and to create buffers for populations occurring within protected areas.

The collection of sighting data should be continued to document population trends and to gauge the efficacy of conservation efforts. It is recommended that these methods be employed in other range states to permit more accurate estimation of national population sizes. Reliable estimates of the distribution and status of large carnivores represent a vital prerequisite for conservation efforts, but are frequently lacking due to the costs associated with sampling over large areas (Smallwood & Fitzhugh 1995). Sightings collection represents an inexpensive and effective way in which the status of a highly visible large predator species can be assessed over large areas (Gros et al. 1996), and has been used to assess the status of several species (e.g. ocelots *Leopardus pardalis*, Tewes & Everett 1986; cougars, Pike et al. 1999; and wolves, Lariviere et al. 2000). The methods have particular applicability to developing countries with limited resources to respond to declines in carnivore populations. Future research should focus upon testing the validity of the methods.

In conclusion, a small population of wild dogs occurs outside state protected areas in South Africa, primarily in the game ranching areas of the extreme north and northeast. Although wild dogs persist at higher human densities and inhabit a wider geographic distribution than previously believed, they remain few in number and inhabit a fraction of the potentially suitable available habitat. Conservation efforts are required to increase the numbers and geographic range of wild dogs in South Africa, and in so doing protect the integrity of populations occurring inside protected areas.

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CHAPTER 3

Area and prey requirements of wild dogs *Lycaon pictus* under varying habitat and land use conditions: implications for reintroductions

3.1 Introduction

Habitat loss is the most significant factor behind ongoing global species extinction (Fahrig 2001). Increasing land use competition for remaining habitat, coupled with insufficient funding for conservation necessitates the conservation of maximal species diversity in minimum areas (Gurd et al. 2001; Restani & Marzluff 2002). Effective conservation planning for activities such as reserve design and endangered species reintroductions is dependent upon understanding species' minimum area requirements. Area considerations are of particular importance to the conservation of large carnivores, whose large spatial requirements have resulted in their being disproportionately affected by habitat loss, and correspondingly difficult to conserve (Linnell et al. 2001). This is complicated by the fact that successful carnivore restoration entails not only the reintroduction of the species, but also the restoration of the ecological relationships between predator and prey and between the predators (Pyare & Berger 2003). Reserve size extinction thresholds for large carnivores are high, but extremely variable. Snow leopards *Panthera uncia* (>116 km^2) and tigers *Panthera tigris* (>135 km^2) for example, require relatively small reserves for persistence, while brown bears *Ursus arctos* (>3,981 km^2 require large areas for persistence (Woodroffe & Ginsberg 1998).

Across all ecosystems wild dogs occur at low densities relative to competing carnivores (Creel & Creel 2002) and are affected by substantial edge effects in all but the largest reserves, as a result of their ranging behaviour (Woodroffe & Ginsberg 1998). It has been suggested that the long-term viability of wild dog populations and the ecological processes that characterise them may require protected areas as large as 10,000 km^2 (Woodroffe & Ginsberg 1999). Using Vortex modelling techniques however, Mills et al (1998) showed that single packs representing sub-populations within a meta-population could be maintained at desirable levels given realistic levels of manipulation through management. In South Africa, the decision was taken to establish a meta-population through the reintroduction of wild dogs into geographically isolated reserves, linked through management, to complement the single viable population occurring in Kruger National Park (henceforth referred to as "Kruger"). Six sub-populations have been established to date: Hluhluwe-Umfolozi Park (960 km^2); Karongwe Game Reserve (120 km^2); Madikwe Game Reserve (750 km^2); Marakele National Park (900 km^2); Pilanesberg National Park (550 km^2); and Venetia Limpopo Game Reserve (370 km^2). Effective predator proof fencing has reduced edge effects and dispersal from the release sites, and enabled the utilisation of reserves smaller than protected areas in which naturally occurring wild dogs persist (550 km^2, Woodroffe & Ginsberg 1998). However, due to high human population densities and intensive agriculture, few suitable state or privately owned areas of this size exist in South Africa, despite the recent increase in the number of areas where wildlife rather than livestock farming is the major form of land use (Falkena 2000). For expansion of the meta-population, the reintroduction of wild dogs into yet smaller areas, as has recently been carried out in the 120 km^2 Karongwe

Game Reserve, could be considered, although in the long term, the formation of large collaborative nature reserves through the cooperation of neighbouring private land owners, is more ecologically desirable.

Several factors may contribute to the large home range areas of wild dogs under natural conditions, including predation by lions *Panthera leo*, interference competition by spotted hyaenas *Crocuta crocuta*, and human-related mortality (Creel & Creel 1998; Mills et al. 1998; Vucetich & Creel 1999). Although each of these factors can be controlled through management to some extent, it is not clear whether wild dogs can be successfully maintained in areas smaller than naturally occurring home ranges. Wild dogs appear to be rarely limited by food availability (Creel & Creel 1998) despite a positive correlation between wild dog density and prey density across ecosystems (Fuller et al. 1992). If wild dog reintroduction is attempted in areas smaller than observed home range sizes, however, prey availability may become a limiting factor. In this study, the minimum areas required to support packs of varying sizes are estimated for different habitat types, based upon prey requirements. The purpose of this is to provide guidelines for minimum area requirements for wild dog reintroductions, and to provide seed values for the adaptive management of sub-populations post-release.

3.2 Methods

The prey population size required to provide a maximum sustainable yield equal to the number of individuals of a given prey species killed by wild dogs in a year was determined, using the following equation, derived from Caughley (1977):

$$MSY = \frac{r_m}{2} \times \frac{K}{2} = \frac{r_m K}{4}$$

Where *MSY* is the maximum sustainable yield, r_m the intrinsic growth rate of the prey population, and *K* the carrying capacity for the prey population. *MSY* becomes N_{prey}, the number of individuals of a prey species killed per year by a pack of wild dogs, and *K* becomes N_{min}, the minimum population size required to support the predation by a pack of wild dogs of a given size over a year:

$$N_{min} = \frac{4N_{prey}}{r_m}$$

The area required to support N_{min} for a given prey species was determined by multiplying N_{min} by the density at which that prey species occurs in a given area.

The following parameters are required to estimate minimum area requirements:

a) Pack size. Estimates were made for the prey requirements of a range of pack sizes (4 - 26 dogs).

b) An estimate of the likely annual increase in wild dog numbers following a reintroduction event. There is likely to be a lag between the birth of additional wild dogs and management action to maintain the desired numbers in a reserve. Subsequently, prey requirements were calculated for a given pack size and one set of offspring. Demographic patterns in packs following reintroductions in South Africa have been extremely variable (e.g. Maddock 1999) with no consistent patterns. Consequently, to be conservative, published (high) survival rates were used to estimate the potential increase in wild dog numbers following one set of offspring. Given average pack structure for Kruger as a whole (five adults and two sub adults), mean litter size (nine pups), and good survival rates (0.8 for adults, 0.7 for sub adults and 0.7 for pups), an initial pack size of seven adult and sub adult dogs would be expected to increase to ~ 12 dogs within the first year (Fuller et al. 1992), as follows:

$$PSt+1 = (Ad \times 0.8) + (S.ad \times 0.7) + (Pt+1 \times 0.7)$$

Where *PSt*+1 is equal to pack size at the end of year 1, where *Ad*, and *S.ad* are equal to adults and sub adults at the beginning of year one respectively, and where *Pt*+1 is equal to the litter of pups born during year one.

c) An estimate of the likely post-release prey-profile of wild dogs for a given area. Documented prey-profiles from three ecosystems: 1) southern Kruger (Mills & Gorman 1997); 2) Save Valley Conservancy in southeastern Zimbabwe (Pole 1999); and, 3) Hluhluwe-Umfolozi Park in Kwa-Zulu Natal (Kruger et al. 1999), were used as approximations of the likely prey-profile of wild dogs in the three areas in which reintroductions are most likely to occur in South Africa: a) the Lowveld in Mpumalanga, and Limpopo (northeastern South Africa); b) northern Limpopo (northern South Africa); and c) northern Kwa-Zulu Natal (eastern South Africa), respectively. Detailed age and sex breakdowns were unavailable for the eastern and northeastern South Africa prey profiles, and consequently standard unit mass was used for each prey species (Coe et al. 1976). From an estimate of the total biomass of prey killed per year by a pack of a given size, the proportion of this made up by each prey species, and the standard unit mass for each species, the number of individuals of each prey species expected to be killed per year by a pack of a given size was calculated. Minimum area estimates were based upon the two dominant prey species in each prey-profile - impala *Aepyceros melampus* and kudu *Tragelaphus strepsiceros* in northeastern and northern South Africa, and nyala *Tragelaphus angasi* and impala in eastern South Africa (Table 3.1).

d) An estimate of the annual biomass of prey killed by a pack of a given size. Adult male wild dogs require a food consumption rate of 3.04 kg / day (Nagy 2001), from which the daily requirements of an average sized individual were estimated, based upon 0.75 mean adult mass for wild dogs (Coe et al. 1976). As a rule, 61% of the body mass of ungulates is made up of flesh (Blumenschine & Caro 1986) and based on this, the daily food requirement estimate was adjusted to yield an estimate of prey biomass killed / dog / day (3.2 kg), approximating to field estimates of 1.8 - 3.5 kg / dog / day (Fuller & Kat 1990; Mills & Biggs 1993; Creel & Creel 1995).

e) An estimate of the density of prey species in the relevant area. Density estimates of the prey species in the three prey profiles considered were taken from census data in each area (southern Kruger - Mills & Gorman 1997; Hluhluwe - KZN Wildlife unpubl. data 1998; Save Conservancy - Pole 1999).

Table 3.1 Percent biomass made up by each prey species in the diet of wild dogs in three ecosystems

Prey species	Eastern SA [a]	Northeastern SA [b]	Northern SA [c]
Bushbuck *Tragelaphus scriptus*	0	2.0	1.2
Cattle *Bos spp.*	0	0	1.0
Grey duiker *Sylvicapra grimmia*	0.1	4.4	0.4
Eland *Taurotragus oryx*	0	0	0.2
Impala *Aepyceros melampus*	16.3	**81.0**	**61.0**
Kudu *Tragelaphus strepsiceros*	0.8	8.1	36.0
Nyala *Tragelaphus angasi*	**76.1** [d]	0	0.2
Red duiker *Cephalophus natalensis*	0.3	0	0
Reedbuck *Redunca arundinum*	0.9	2.0	0
Steenbok *Raphicerus campestris*	0	2.5	0
Waterbuck *Kobus ellipsiprymnus*	2.0	0	0
Wildebeest *Connochaetes taurinus*	3.5	0	0
Total	100	100	100

[a] Kruger et al. (1999).
[b] Mills & Gorman (1997).
[c] Pole (1999).
[d] Bold figures highlight the dominant prey species in the wild dog prey-profile.

f) An estimate of the intrinsic growth rate (r_m) of each prey species. This was estimated as follows (Caughley & Krebs 1983):

$$r_m = 1.5W^{-0.36}$$

W represents the mean adult live body mass (Bothma 1996; Baker 1999).

Calculation of the minimum area requirements of wild dogs in this fashion assumes: 1) that the prey-profile of wild dogs in small protected areas will be similar to that observed in large protected areas; 2) that the numerical impact of wild dogs upon prey populations is not influenced by age or sex-based prey selection; 3) that carrying capacity remains constant for prey populations, and 4) that the prey populations in the three reference ecosystems were stocked at ecological carrying capacity.

3.3 Results

The estimated minimum area required to support wild dogs is greatest in northeastern South Africa, followed by eastern South Africa (Table 3.2). An estimated minimum area of 354.2 km^2 is required to support an average pack of seven adult and sub adult dogs, and one set of offspring (12 dogs in total) in northeastern South Africa, compared to 172.8 km^2 in eastern South Africa, and 158.5 km^2 in northern South Africa. Nyala in eastern South Africa, impala in northeastern South Africa and kudu in northern South Africa set the greatest minimum area requirements.

Table 3.2 Minimum population sizes and areas required to support predation by a pack of 12 wild dogs (pack of seven dogs, plus one year's offspring at one year of age), given three prey-profiles

Ecosystem / Species	N_{prey}[a]	r_m[b]	N_{min}[c]	Prey density / km^2	Area required (km^2)
Eastern SA					
Impala	56	0.38	591	0.09	65.4
Nyala	144	0.30	1904	0.11	172.8
Northeastern SA					
Impala	281	0.38	2950	0.08	354.2
Kudu	8	0.23	143	0.008	173.1
Northern SA					
Impala	211	0.38	2223	0.15	148.4
Kudu	37	0.23	638	0.04	158.5

[a] Estimated number of individuals of a prey species killed per year by a pack of 12.
[b] Estimated intrinsic growth rate of the prey population.
[c] Estimated minimum prey population size required to support predation by a pack / year.

A pack size of five adults represents the statistical threshold above which pack survival is likely, and below which pack extinction is likely (Courchamp & Macdonald 2001). The predicted minimum area requirements for a pack of five wild dogs in northern South Africa are comparatively low at 130.8 km^2 (given five adults and one set of offspring - 10 dogs in total, Figure 3.1). In comparison, estimates for eastern and northeastern South Africa are higher, at 144.3 km^2 and 294.6 km^2, respectively. At large pack sizes, the predicted minimum area requirements are much higher. In northern South Africa, 18 wild dogs are predicted to require a minimum area of 235.6 km^2, compared to 259.7 km^2 in eastern South Africa, and 530.0 km^2 in northeastern South Africa.

Observed home range areas are 1.2, 2.0 and 3.5 times larger than the estimated minimum areas required to support a pack equal to the mean pack size observed in eastern, northeastern and northern South Africa, respectively (Table 3.3). In keeping with this, the density at which wild dogs occur in the three reference areas is markedly lower than the theoretical maximum potential density that each area could support, if wild dogs were regulated by density dependent resource limitation (Table 3.4). The observed density of wild dogs is 0.21 that of the maximum potential density in eastern South Africa (taking the density of dogs within Hluhluwe-Umfolozi Park as a whole), 0.31 that of the potential density in northern South Africa, and 0.61 of potential density in northeastern South Africa.

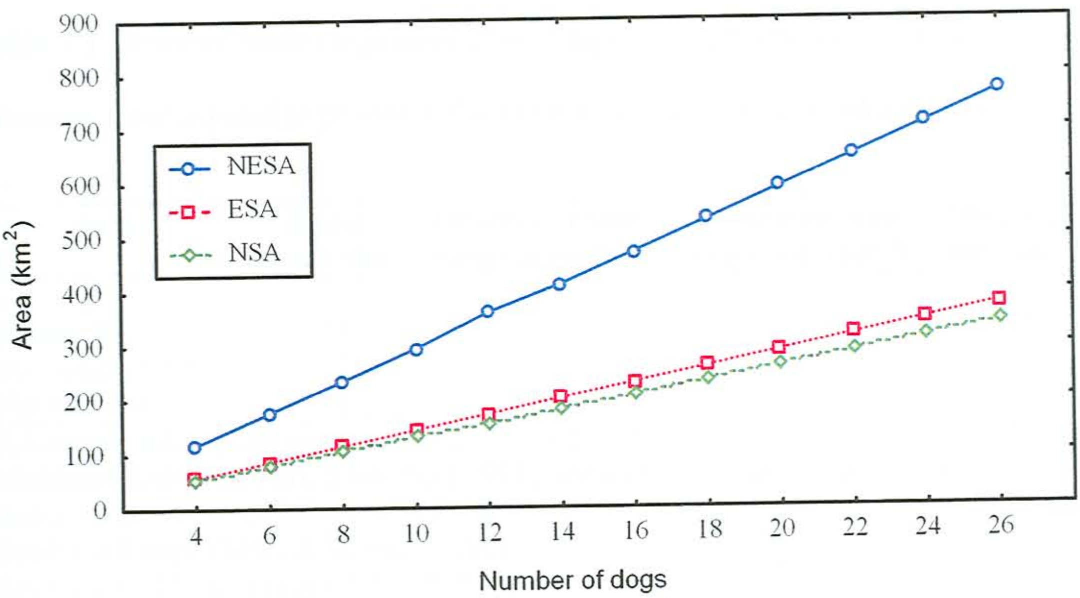

Figure 3.1 Minimum areas required to support predation by varying pack sizes, based upon the dominant prey species in three different prey profiles (ESA, eastern South Africa - nyala; NESA, northeastern South Africa - impala; NSA, northern South Africa - kudu)

Table 3.3 Observed home range areas of wild dogs in three ecosystems, versus estimated minimum areas required to provide sufficient prey to support equivalent pack sizes

Area	[a] Mean pack size	Observed home range areas (km^2)	Estimated area required (km^2)	Observed : estimated
[b] Eastern SA	13	218	188	1.2
[c] Northeastern SA	9	537	265	2.0
[d] Northern SA	9	414	117	3.5

[a] Adults, sub adults, and number of pups divided by 2, after Mills & Gorman (1997).
[b] Hluhluwe-Umfolozi Park (December 1994 pack size – Maddock 1999, 1993 - 1994 home range size - Andreka et al. 1999).
[c] Southern Kruger (Mills & Gorman 1997).
[d] Save Valley Conservancy (Pole 1999).

Table 3.4 Observed density of wild dogs in three ecosystems, versus estimated maximum density (dogs / 100 km^2) at which wild dogs would occur if they were regulated by density dependent resource limitation across three ecosystems

Area	Observed density	Estimated maximum density	Observed : Estimated
Eastern SA [a]	1.44	6.93	0.21
Northeastern SA [b]	2.07	3.39	0.61
Northern SA [c]	2.40	7.64	0.31

[a] Hluhluwe-Umfolozi Park (1994 density data, Maddock 1999).
[b] Southern Kruger (Maddock & Mills 1994).
[c] Save Valley Conservancy (Pole 1999).

3.4 Discussion

The validity of minimum area estimates presented in this chapter is dependent upon the validity of the underlying assumptions. It was assumed that the prey-profile of wild dogs reintroduced into small areas would approximate to that observed in large areas of similar habitat. Wild dogs usually prey upon the most abundant medium to large prey species and typically take prey in proportion to abundance (Reich 1981; Fuller & Kat 1990; Pole 1999), suggesting that approximate prey-profiles can be predicted. Experience from recent reintroductions, however, suggests that wild dogs reintroduced into small areas learn to utilise perimeter fencing during hunting, enabling the capture of larger prey (Hofmeyr 1997; van Dyk & Slotow 2003). There is also the possibility that in the short term after a reintroduction, naïve prey will be more susceptible to predation (Hunter 1998). Given this scope for variation in prey-profiles, monitoring prey population trends as well as prey selection following a reintroduction is important to guide the regulation of wild dog or prey numbers in line with management objectives and ecological conditions.

It was also assumed that prey populations in the reference areas were at carrying capacity and that prey population sizes were estimated correctly. Estimates of wildlife densities are often inaccurate (Bell 1986) and liable to underestimate numbers of small species such as impala (Creel & Creel 2002). In addition, prey densities used to derive minimum area estimates represent densities after off-take by a large predator guild, including wild dogs. Both factors are likely to result in conservative minimum area estimates.

In estimating minimum required prey populations, no consideration was made for sex or age selection by wild dogs. The effect of this is likely to be further conservatism in minimum area estimates. Mills & Shenk (1992) showed that lion predation had a lower impact upon zebra *Equus burchelli* than wildebeest *Connochaetes taurinus* populations in Kruger, as a result of selection for juveniles in the former species. In keeping with this, wild dogs select for juveniles when hunting larger species (Kruger et al. 1999; Pole 1999; Creel & Creel 2002) and as a result, are likely to have a lower impact on populations of these species. Furthermore, wild dogs select for impala in poor condition (Pole 1999, Pole et al. in prep.), and subsequently, a portion of predation by wild dogs may compensate for animals that would have died anyway.

It is also assumed that carrying capacities for prey populations are constant. In reality, however, carrying capacities vary continuously and markedly with environmental conditions (Bell 1986). Ungulate numbers are likely to drop during times of drought, and wild dog numbers are likely to increase due to improved conditions for hunting (Mills 1995). During drought in Kruger between 1981 - 1983 for example, impala and kudu populations declined by 30 - 40% (Walker et al. 1987), and in a drought in the early 1990s, wild dog numbers increased (Mills 1995). Despite reduced prey availability, weakening caused by food shortages in a drought is likely to make alternative prey species available to wild dogs, and improve hunting success (Mills 1995). In addition, the logistical model used to determine minimum required prey population sizes is conservative and likely to cater for some variation in carrying capacity (Caughley & Sinclair 1994). In wet years, ungulate numbers are likely to increase, and wild dog numbers might be expected to decrease, as occurred in Kruger between 1995 - 2000, probably due to poor conditions for hunting (Davies 2000). These patterns stress that if reintroduction into a small area is carried out, it is necessary to monitor wild dog and prey numbers to evaluate population trends, to learn how and when to manage these processes. One way that this could be accomplished would be to set limits of acceptable change, although these are always subjective.

The results of this chapter suggest that in the absence of other factors, smaller areas than those typically considered for wild dog reintroductions provide sufficient prey resources to support wild dog packs. Supporting this, wild dogs reintroduced into three reserves in South Africa have utilised smaller home range areas than typically recorded in large protected areas (Fuller et al. 1992): Hluhluwe-Umfolozi Park (218 km^2, Andreka et al. 1999); Pilanesberg National Park (13.4 km^2 during the first 8 months post-release, van Dyk & Slotow 2003); and Madikwe Game Reserve (180 km^2, Hofmeyr 1997). My study suggests that in areas of high prey density, protected areas as small as 130 km^2 have the potential to support a small pack of wild dogs and one year's offspring. Reserves of this size are comparatively common in South Africa (Chapter 4, van Dyk & Slotow 2003), suggesting that the number of sites potentially available for wild dog reintroduction might be greater than previously thought (Mills et al.1998). Given the dynamic nature of wild dog packs and other aspects of their ecology (Creel and Creel 1998), the practical application and management of this needs to be studied in more detail.

The predicted maximum density at which wild dogs could occur in northeastern South Africa given density dependent resource limitation (3.39 dogs / 100 km^2), is markedly lower than the observed density of some other large carnivores (approximately 10 lions and spotted hyaenas / 100 km^2, Mills & Gorman 1997) in the same habitat. This indicates that wild dogs require large areas simply to meet prey requirements. Wild dogs have extremely high rates of daily energetic expenditure (Gorman et al. 1998; Nagy 2001) and consequently high daily food consumption rates (Fuller & Kat 1990; Creel & Creel 1995; Fuller et al. 1995). Furthermore, in contrast to lions and spotted hyaenas, wild dogs do not exploit carrion, utilise a narrow range of prey species (Ginsberg & Macdonald 1990; Fuller et al. 1992; Mills & Biggs 1993), and because of their cursorial hunting technique, appear to be selective for individuals (Pole 1999; Pole et al. in prep.). Wild dogs are rate-maximisers (Kruger et al. 1999; Pole 1999) and it is possible that they range widely in order to attain a constant food supply with minimal effort (Schaller 1972).

However, prey availability explains little of the observed variation in wild dog density between protected areas (Creel & Creel 1998), and wild dogs occur at lower densities than predicted by body mass and available prey biomass (Carbone & Gittleman 2002). Therefore, beyond a threshold of prey availability, other factors are likely to influence wild dog density (Pole 1999). In keeping with this, for the three reference areas considered, observed densities are 2 - 5 times lower than the theoretical maximum densities based on prey availability. Management of the factors responsible for this discrepancy has potential to enable the maintenance of wild dogs at densities closer to the theoretical maximum.

Across ecosystems, there is a negative correlation between the density of wild dogs and the density of lions and spotted hyaenas (Creel & Creel 1996). Spotted hyaenas affect wild dogs through interference competition, and lions are significant agents of mortality (Creel & Creel 1996; Mills & Gorman 1997), causing an estimated 33% of mortality in Kruger (van Heerden et al. 1995). In the presence of intact predator guilds, wild dogs avoid areas of high prey density as a mechanism to avoid high densities of lions (Mills & Gorman 1997; Creel & Creel 2002). In contrast, in an area with low densities of lions, wild dogs were observed to select habitat in a pattern consistent with prey availability (Pole 1999). Lions probably reduce wild dog density through direct mortality and by reducing their access to optimal habitats. In small-protected areas, the potential for spatial niche differentiation between competing predator species is reduced, and subsequently, lion and spotted hyaena numbers are likely to require intensive management to enable wild dogs to persist. It has been suggested that reducing the size of male lion coalitions can increase the success of wild dog reintroductions, by reducing the area covered during the movements of these lions, thus providing more scope for the wild dogs to avoid contact with them (van Dyk & Slotow 2003).

Human-related wild dog deaths represent a major source of mortality in some populations (Creel & Creel 1998) and may limit density in some areas. Most human-related deaths occur as a result of wild dogs leaving protected areas (Woodroffe & Ginsberg 1997) and the maintenance of perimeter fencing has the potential to dramatically reduce this source of mortality. In addition, anti-poaching management and strict speed limits within a reserve are likely to reduce mortality due to snaring and road accidents.

Finally, disease has had a significant impact upon some wild dog populations (Creel & Creel 1998). Although, typically episodic in nature, infectious diseases cause 11% of adult wild dog mortalities across ecosystems (Woodroffe & Ginsberg 1997). Vaccination of wild dogs prior to reintroduction is vital to reduce mortality and to avoid catastrophic outbreaks such as that experienced following the first reintroduction attempt at Madikwe Game Reserve in South Africa (Hofmeyr et al. 2000).

Although it is difficult to determine what is an adequate reserve size for reintroductions (Miller et al. 1999), the results presented in this chapter give estimates and provide seed values for adaptive management of wild dogs and their prey in line with reserve management objectives. Estimating minimum required reserve size is a major focus of conservation biology (Rodrigues & Gaston 2001; Wieglus 2002). Most research has considered the minimum areas required for viable populations, and comparison between the area requirements of wild dogs and most other large carnivore species highlights the difficulties associated with wild dog conservation. Beier (1993) estimated that extinction risk in cougars *Felis concolor* in southern California is low in areas as small as 2,200 km^2, while Sconewald-Cox et al. (1988) estimated that the minimum area required to conserve a viable population of wolves *Canis lupus* indefinitely is as low as 1,080 km^2. By comparison, it has been suggested that wild dogs may require areas as large as 10,000

km^2 for long-term viability (Woodroffe & Ginsberg 1999). Correspondingly, the critical minimum reserve size below which wild dog extinction is predicted (3,606 km^2) is substantially larger than that predicted for their competitors and ecological equivalents (lions – 291 km^2; spotted hyaenas – 179 km^2; wolves - 723 km^2, Woodroffe & Ginsberg 1998).

Increasing habitat loss is likely to increase the need for utilising fragments of natural land cover, and employing meta-population management techniques (Griffith et al. 1989). The methods outlined in this chapter are applicable to the conservation of minimum demographic units or viable populations of any endangered carnivore in remaining habitat fragments. Applicability is perhaps greatest in southern Africa, where the prevalence of fenced reserves and other game areas permits the conservation of large carnivores in small habitat fragments.

In conclusion, the methods developed in this chapter provide a means by which to determine minimum areas required for the reintroduction of wild dogs, or other endangered carnivore species. Given management to reduce the negative impact of other factors upon wild dogs, smaller areas than previously considered may represent potential reintroduction sites.

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CHAPTER 4

Attitudes of ranchers towards African wild dogs *Lycaon pictus*: conservation implications on private land

4.1 Introduction

Due to their position at the top of the food chain, large carnivores (>5 kg) are less common than their prey species, require large areas with sufficient prey, and tend to come into conflict with human interests, creating unique problems for their conservation (Sillero-Zubiri & Laurenson 2001). Spatial and temporal variation in the persistence of large carnivore species is primarily a function of variation in human willingness to tolerate, and ability to kill such species (Woodroffe 2000). State sponsored eradication campaigns caused the extirpation of several large carnivore species from major portions of their natural range (Breitenmoser 1998; Berg 2001). For example, during the late 1800s and early 1900s federal money was used to support professional hunters using traps, poisons and aerial hunting to kill carnivore species such as wolves *Canis lupus* and cougars *Felis concolor* in North America (Linnell et al. 2001). Today, persecution by humans is still the greatest source of mortality for many large carnivore species occurring both outside and inside protected areas (Woodroffe & Ginsberg 1998).

Variation in attitudes of people towards large carnivores appears to be based partly upon the extent to which different species conflict with human interests, and partly upon inherent human prejudices (Kellert 1985). Wild canids, in particular, seem to engender dislike. In North America, although bears *Ursus arctos* are traditionally popular and perceived to be 'worthy' of conserving, wolves and coyotes *Canis latrans* are among the least liked animals (Kellert 1985; Berg 2001). In Africa, wild dogs have suffered from similarly negative perceptions and fare poorly in the public eye relative to other species (Fanshawe et al. 1991). A killing method sometimes used – ripping at the abdomen to disembowel prey has resulted in wild dogs being described as 'cruel', while other misconceptions include the suggestion that wild dogs regularly kill more than they need, and that they 'terrorise' their prey (Mills & Nel 1993; Woodroffe & Ginsberg 1999). In the past, naturalists have further contributed to negative perceptions by using phrases such as 'murderous', 'destructive', 'rapacious', 'abomination', 'unnecessary creature', 'hound of hell', 'killing wantonly', 'far more food than they need', and 'utmost cruelty' to describe wild dogs and their behaviour (Maugham 1914; Bere 1955; Hunter 1960; Alexander 1986). Antagonism towards wild dogs was reflected in state policies in some countries, and they were killed as vermin in protected areas in at least four southern African nations until as late as 1975 (Creel & Creel 2002). Although state-sponsored persecution has ceased, many of the myths surrounding wild dogs persist, and promote continued persecution of the species outside protected areas. Rasmussen (1999), for example, contends that ranchers have "an intrinsic loathing of wild dogs" and put "prejudice before rationality".

Several life history traits predispose wild dogs to conflict with humans. First, they are wide ranging and so packs are liable to move out of protected areas (Mills 1991;

Woodroffe & Ginsberg 1998), partially because they avoid areas with high densities of lions *Panthera leo* (Mills & Gorman 1997). Secondly, they are diurnal, highly visible, and relatively fearless, increasing the likelihood of lethal encounters with humans (Frank & Woodroffe 2001). Finally, their obligate cooperative breeding system is vulnerable to an Allee-effect at low pack sizes (Courchamp & Macdonald 2001), which increases the impact of persecution by humans.

Persecution by humans in conjunction with habitat destruction has been responsible for a major reduction in the numbers and geographic range of wild dogs, and the current world population is estimated to number as few as 3000 individuals (Fanshawe et al. 1997). In South Africa, range reduction was particularly marked, and wild dogs are presently restricted to a single viable population occurring in the Kruger National Park (henceforth referred to as "Kruger"). Current conservation efforts in South Africa have focused upon the creation of a meta-population through the reintroduction of wild dogs into a series of isolated reserves (Mills et al. 1998). Prior to the establishment of the proposed transfrontier parks, future expansion of the distribution of wild dogs in South Africa is likely to depend increasingly upon reintroductions into privately owned reserves, and through the conservation of naturally occurring wild dogs *in situ* on ranchland, due to a relative shortage of suitable state-protected areas.

A shift in land use patterns across much of southern Africa from cattle ranching to game ranching has created large areas of potentially suitable habitat for wild dog conservation. In parts of Zimbabwe, wild dogs have re-colonised parts of their former range on private land (Pole 1999; Rasmussen 1999), and resident packs have become established on private land in several parts of South Africa (Chapter 2). The success of conservation efforts involving wild dogs or other carnivores outside state-protected areas is, however, entirely dependent upon the willingness of people to tolerate their presence.

Understanding the attitudes of private land managers towards wildlife is a vital step in the establishment of conservation projects (Newmark et al. 1994), and in North America for example, several studies have gauged opinions prior to the onset of carnivore conservation initiatives (Lohr et al. 1996; Pate et al. 1996). In this study the attitudes of southern African ranchers towards wild dogs were assessed to determine the conditions under which conservation initiatives might succeed and to identify ways in which conflict might be reduced.

4.2 Methods

Sampling was conducted in three parts of South Africa, and three parts of Zimbabwe in which wild dogs are known to occur on private land (Childes 1988; Skinner & Smithers 1990; Fanshawe et al. 1997; Maddock 1999; Pole 1999), yielding a total sample of 211 ranchers. In South Africa, rancher's attitudes were sampled at the following sites (approximate central co-ordinates in parentheses): northern Kwa-Zulu Natal (n = 26, 27° 30' S, 31° 45' E); the western Kruger border (n = 82, 24° 10' S, 30° 55' E); and the Limpopo Valley (n = 58, 22° 20' S, 29° 40' E); hereafter referred to as eastern, northeastern and northern South Africa, respectively. In Zimbabwe, rancher's attitudes were sampled in the Save Valley Conservancy (n = 15, 20° 05' S, 32° 00' E), in and adjacent to the Gwayi River Conservancy (n = 19, 18° 40' S, 27° 10' E), and in the Matetsi ranching area (n = 11, 18° 26' S, 26° 07' E).

In South Africa, focal areas of wild dog activity were demarcated with the assistance of nature conservation representatives, and a list of ranch names obtained for each site from 1 / 250 000 maps. Variation in sample sizes between sample sites reflects variation in the number of ranches lying within the demarcated focal areas of wild dog activity. Contact details for ranchers were derived from telephone directories. In each area, as many ranchers as possible were interviewed in a two-week period. In Zimbabwe, ranching communities were smaller and an attempt was made to contact all of the ranchers in each area.

At all sample sites, ranch owners or managers were interviewed in person, with the exception of the few cases (<5%) where ranches were too remote for access by car, in which case telephonic interviews were conducted. Respondents were informed that the University of Pretoria was conducting the project, and assured that all responses would remain anonymous. Universities are often seen as a neutral body, encouraging honesty and reducing compliance bias (Mitchell & Carson 1989). The refusal rate was <3%.

Respondents were interviewed with a structured questionnaire (Appendix B). Pre-testing was conducted on ranchers in Zimbabwe prior to the study, to ensure that all questions were clear and understandable, and a final version was prepared for sampling. The questionnaire was divided into three components. (1) a 'Ranch Characteristics' section, concerning ranch characteristics relevant to wild dogs: property size; fencing characteristics; land use; severity of poaching and whether or not the ranch was part of a collaborative nature reserve. Collaborative Nature Reserves (CNRs) or conservancies are private nature reserves composed of multiple adjacent properties with internal fencing removed, and surrounded by a single perimeter game fence (Lambrechts 1996). (2) A 'Predators' section, concerning the occurrence of, and attitudes towards six species of mammalian carnivore. (3) A 'wild dogs' section concerning the occurrence of wild dogs and attitudes towards them.

4.2.1 Statistical analysis

The relationship between rancher's attitudes and species (black backed jackals *Canis mesomelas*, cheetahs *Acinonyx jubatus*, leopards *Panthera pardus*, lions, spotted hyaenas *Crocuta crocuta* and wild dogs) was investigated using ordinal logistic regression, based upon answers to question 12 (Appendix B). Attitudes towards each species (question 12, Appendix B) were categorised as negative (scores 0 - 1), neutral (scores 2 - 3), or positive (scores 4 - 5). The relationship between rancher's attitudes towards wild dogs, and four ranch characteristics was also analysed using ordinal logistical regression. Attitude towards wild dogs, as the dependent variable, was categorised as negative or positive based upon responses to question 17 (Appendix B). Four categorical independent variables were included. 1) Geographic region - ranches were categorised as eastern, northeastern or northern South African, or Zimbabwean. The Zimbabwean ranches were included as a single category due to low sample sizes. 2) Part or not part of a CNR -

ranches were classified as being isolated, or part of a CNR. 3) Ranch size - ranches were categorised as small (0 – 1,450 ha), medium (1,451 – 4,200 ha) or large (≥4,201 ha), yielding three categories with approximately equal numbers of samples. 4) Land use - ranches were categorised into four groups on the basis of land use: a) cattle; b) cattle / consumptive wildlife utilisation; c) consumptive wildlife utilisation alone; and d) where ecotourism was a land use (alone or in conjunction with consumptive wildlife utilisation), or where the maximisation of economic benefits was not a priority.

4.3 Results

Mean ranch size (± S.E.) was 3,290 (± 742) ha in eastern South Africa, compared to 3,185 (± 529) ha in northern South Africa, and 2,865 (± 497) ha in northeastern South Africa. In South Africa, ranches were typically surrounded by perimeter game fencing (Table 4.1), and in eastern (43.8% electrified, 55% meshed), and northeastern South Africa (51.3% electrified, 3.0% meshed), fencing was frequently electrified or meshed, both of which reduces access to wild dogs (Hofmeyr 2000). In northeastern and eastern South Africa 48.2% and 23.0% of ranches belonged to CNRs, compared to 0% in northern South Africa. CNRs varied in size from 2,500 – 65,000 ha in northeastern South Africa, and 4,500 – 20,000 ha in eastern South Africa. Consumptive utilisation of wildlife was prevalent at each South African site. Livestock ranching was common in northern (39.0% of ranches had livestock) and eastern South Africa (49.9% of ranches), while ecotourism was common in both northeastern (32.5% of ranches) and eastern South Africa (27% of ranches, Table 4.2).

Table 4.1 The percentage of ranches with various fencing characteristics (number of ranches in parentheses)

	ESA [a] (n=26)	NESA [b] (n=82)	NSA [c] (n=58)	Gwayi [d] (n=19)	Matetsi [d] (n=11)	Save [d] (n=15)
No fencing	0	0	1.8	15.8	38.0	5.3
Perimeter cattle fencing	15.0	0	12.7	15.8	18.0	0
Partial perimeter game fencing	25.0	16.0	10.9	68.4 [e]	0	94.7 [e]
Perimeter game fencing	60.0	84.0	76.4	0	45.0	0
Electrification	43.8	51.3	12.0	0	27.2	0
Mesh fencing	55.0	3.0	2.4	0	0	0
Ranches part of CNRs [f]	23.0	48.2	0	94.7	0	100

[a] Eastern South Africa.
[b] Northeastern South Africa.
[c] Northern South Africa.
[d] The Zimbabwean sample sites.
[e] The outer perimeter fence of a large collaborative nature reserve.
[f] Collaborative nature reserves.

Table 4.2 The percentage of ranches with various land uses (number of ranches in parentheses)

Land use	ESA [a] (n=26)	NESA [b] (n=82)	NSA [c] (n=58)	Gwayi [d] (n=19)	Matetsi [d] (n=11)	Save [d] (n=15)
Cattle	3.9	17.5	26.0	0	9.1	0
Cattle / CWU [e]	46.0	11.2	13.0	10.6	0	0
CWU	19.2	10.0	31.5	31.5	54.5	46.7
CWU / ecotourism	3.9	11.2	18.5	21.1	18.2	53.3
Ecotourism	23.1	21.3	5.5	36.8	18.2	0
None	3.9	28.8	5.5	0	0	0

[a] Eastern South Africa.
[b] Northeastern South Africa.
[c] Northern South Africa.
[d] Zimbabwean sample sites.
[e] Consumptive wildlife utilisation.

Individual ranches were larger in Zimbabwe (11,105 ± 1,370 ha; mean ± S.E.) than in South Africa (3,047 ± 236 ha; Mann Whitney; $T = 6860.5$; $p<0.0001$), with land uses based primarily upon the consumptive utilisation of wildlife, and ecotourism (Table 4.2). CNRs in Zimbabwe were also larger (Save Valley - 360,000 ha; Gwayi River – 92,000 ha; cf. 4,500 – 65,000 ha in South Africa). Ranches belonging to CNRs typically had partial fencing (on the outer boundary of the CNR), whereas Zimbabwean ranches not part of CNRs were often unfenced (50.0% of ranches), with few single ranches having electric fencing (21.4%).

All predators were more commonly seen on Zimbabwean than South African ranches (Table 4.3). Wild dogs were regularly sighted on 91.1% of Zimbabwean ranches, compared to 30.8%, 19.6% and 3.8% of ranches in northeastern, northern and eastern South Africa respectively. Rancher's attitudes towards the six carnivores differed between species ($\mathcal{X}^2 = 51.39$, df = 5, $p<0.0001$). Based upon answers to Question 12 (Appendix B), wild dogs (2.71 ± 0.15; mean score ± S.E.), followed by lions (2.82 ± 0.15) were the least popular species (Figure 4.1). Leopards (3.87 ± 0.12), and black backed jackals (3.57 ± 0.11) were the least unpopular species. Spotted hyaenas (3.24 ± 0.13) and cheetahs (3.35 ± 0.13) were of intermediate popularity. Attitudes towards all predators, and particularly wild dogs and lions, were polarised. For wild dogs and lions, a similar proportion of ranchers denoted scores of zero (very negative: wild dogs 30.7% of ranchers; lions 30.6%) and five (very positive: wild dogs 39.4% of ranchers;

Table 4.3 The percentage of ranches on which various predator species are 'regularly sighted'

Species	ESA [a] (n=26)	NESA [b] (n=82)	NSA [c] (n=58)	Gwayi [d] (n=19)	Matetsi [d] (n=11)	Save [d] (n=15)
Black-backed jackal *Canis mesomelas*	100	91.5	98.2	100	100	100
Cheetah *Acinonyx jubatus*	15.4	58.4	57.1	47.4	81.8	93.3
Leopard *Panthera pardus*	65.4	86.6	78.6	95.1	100	100
Lion *Panthera leo*	11.5	48.2	5.4	78.9	73.2	40.0
Spotted hyaena *Crocuta crocuta*	38.5	79.3	39.3	100	100	73.3
Wild dog *Lycaon pictus*	3.8	30.8	19.6	84.2	81.8	100

[a] Eastern South Africa.
[b] Northeastern South Africa.
[c] Northern South Africa.
[d] Zimbabwean sample sites.

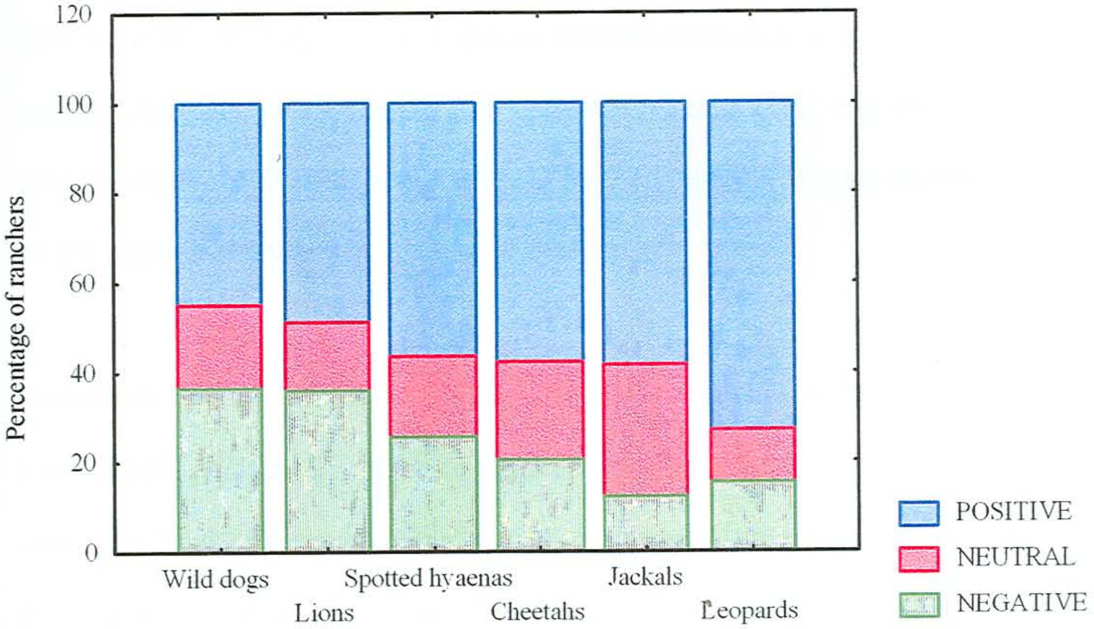

Figure 4.1 Percentage of ranchers who gave negative (scores 0-1), neutral (scores 2-3) and positive (scores 4-5) towards various carnivore species (in response to question 12, Appendix B)

lions 42.1%). The other carnivore species received fewer zero scores (jackals 10.1% of ranchers; leopards 13%; cheetahs 18.6%; spotted hyaenas 21%), with most ranchers denoting scores of five (leopards 62.8% of ranchers; spotted hyaenas 46.4%; cheetahs 44.6%; jackals 41.2%).

In question 12 (Appendix B), ranchers were asked to provide reasons for their attitudes towards each predator. The most common reasons for negative attitudes towards wild dogs were "they affect my income" (13.5% or ranchers), "they kill a lot / too much game" (13%), "they kill livestock" (12%), and "the ranch is too small for them" (10.1%, Table 4.4). Most of these comments were more commonly applied to wild dogs than to any other species. Two comments were made almost exclusively of wild dogs "they chase game and make it wild" (10.6% of ranchers), and "they chase game into fencing during hunting" (6.2%). Overall, 3.7% of ranchers indicated that they would shoot wild dogs on their property (compared to 1.9% - 3.4% for the other species). Discussion with local nature conservation representatives however, suggests that the proportion of ranchers who actually do shoot predators is greater than suggested by the survey results. Interestingly, only 2.3% of ranchers mentioned that they consider the method of hunting used by wild dogs as being cruel, and <1% of ranchers were negative towards wild dogs simply out of a dislike for the species.

The most common reasons for negative attitudes towards cheetahs were "they kill a lot / too much game" (12.5% of ranchers), and "they kill livestock" (6.3%). Almost five percent of ranchers (4.8% of ranchers) complained that cheetahs "waste food", a

Table 4.4 The ten most common reasons for negative and positive attitudes towards six carnivore species

Reasons given for attitudes	Wild dogs	Cheetahs	Jackals	Leopards	Lions	Hyaenas
Negative comments						
They affect income / have no value	13.5	1.9	3.4	2.0	2.9	4.3
They kill a lot of / too much game	13.0	12.5	10.1	2.5	9.1	7.2
They kill livestock	12.0	6.3	6.3	8.7	15.1	11.5
They chase game and make it wild	10.6	1.4	0	0	0	0
The ranch is too small for them	10.1	2.4	0	1.1	9.1	1.0
They chase game into fences	6.2	0	0	0	0	1.0
I will shoot them if I see them	3.7	1.9	1.9	2.4	3.4	1.9
There are too many of them	1.0	0	0	0.5	0	1.0
I don't like them	0.5	0	0	0	0	6.3
They pose a risk to human safety	0	0	0	0.5	5.8	0
They kill for the sake of it / waste meat	0	4.8	0	0	1.1	0
Positive comments						
Their value for ecotourism	21.6	16.3	13.5	22.1	19.6	19.7
Their ecological role / part of the system	13.5	18.3	24.0	13.9	11.6	19.7
Because they are few / only pass through	12.5	5.3	0	4.9	0	4.8
To assist with their conservation	7.2	1.9	0.5	0	1.1	0.5
I like them / they are nice to see	5.8	10.0	2.9	7.7	6.8	4.8
They are no problem / don't kill too much	3.9	12.5	21.0	22.5	3.9	16.8
They are OK if their numbers are managed	3.8	0	9.0	2.9	3.4	0
Their value for trophy hunting	0	0	2.9	8.7	8.2	2.4
They are a valuable species	0	0	0	3.9	3.9	1.0
They make a nice sound	0	0	3.4	0	0	1.9
They clean the bush of carcasses	0	0	7.0	0	0	7.2

complaint rarely levelled at other species. Most common reasons for negative attitudes towards leopards, spotted hyaenas and lions were that "they kill livestock" (8.7%, 11.5%, 15.1% of ranchers, respectively). For lions, 9.1% of ranchers were negative because they felt that the ranch was too small, and 5.8% of ranchers were negative because they felt that lions were a threat to human safety. For spotted hyaenas, 6.3% of ranchers were negative because they "don't like the species", a reason rarely given for negativity towards the other species.

The most common reason given for positive attitudes towards wild dogs was "their value for ecotourism" (21.6% of ranchers), and more ranchers recognised the ecotourism value of wild dogs than of spotted hyaenas (19.7%), lions (19.6%), cheetahs (16.3%), or jackals (13.5%). Fourteen percent (13.5%) of ranchers were positive towards wild dogs because of "their ecological role". Two reasons for positive attitudes towards wild dogs were rarely given to other species: "because they only pass through" (12.5% of ranchers); and "because they are endangered / to assist their conservation" (7.2%).

Ecotourism value and "their ecological role" were common reasons for positive attitudes towards all species, while "their value for hunting" was a reason for positive attitudes towards leopards (8.7% of ranchers) and lions (8.2%). A common reason for positive attitudes towards cheetahs (12.5% of ranchers), spotted hyaenas (16.8%), jackals (21.0%), and leopards (22.5%) was that "they are not a problem / don't kill too much", a reason rarely given for positive attitudes towards wild dogs (3.9%) or lions (3.9%).

Several ranchers were positive towards spotted hyaenas (7.2%) and jackals (7.0%) because "they clean the bush of carcasses".

Ranchers were asked to indicate whether they agree or disagree with a number of statements pertaining to wild dogs: 93.7% of ranchers agreed that "wild dogs are a natural component of a healthy ecosystem"; 12.0% agreed that "wild dogs regularly kill more food than they require"; 61.4% agreed that "wild dogs cause disruption of game herds and make them more skittish"; 41.5% agreed that "wild dogs cause damage to fencing during hunting"; 52.2% agreed that wild dogs are "a liability to a rancher because they consume valuable wildlife but provide no economic return". Although 92.8% of ranchers agreed that "tourists are interested in seeing wild dogs", only 42.7% believe that "sufficient money can be made from marketing 'wild dog eco-tours' to compensate for the losses caused by their predation". Significantly, however, of ranchers who previously indicated that they do not want to have wild dogs on their property, 55.5% indicated that the demonstration of a way in which to derive sustainable profit from wild dogs would be sufficient incentive for them to want the species on their property.

4.3.1 Relationship between attitudes and ranch characteristics

In response to question 17 (Appendix B), 52.3% of ranchers indicated that given a choice they would like to have wild dogs on their property, the remainder stating that they would rather not have wild dogs on their land. Rancher's attitudes varied significantly with variation in several ranch characteristics ($\mathcal{X}^2 = 79.4$, df = 9, p<0.0001), permitting identification of conditions likely to be unfavourable for wild dog conservation, conditions in which wild dog conservation may be possible given public relations exercises and / or wild dog ecotourism schemes, and the conditions likely to be favourable for wild dog conservation without intervention by conservationists (Figure 4.2). Ranchers were most positive ($p=0.0194$) in eastern South Africa (69.2% of ranchers were positive), followed by Zimbabwe (66.6%), northeastern South Africa (58.5%), and northern South Africa (24.1%). In eastern South Africa, 64,537 ha of land sampled (75.5% of total), consisted of land managed by people who wanted wild dogs on their property, compared to 213,176 ha (68.9% of total) in northeastern (excluding three CNRs with open borders with Kruger), and 70,441 ha (39.4% of total) in northern South Africa. Ranchers belonging to a CNR were more positive ($p=0.0065$) than ranchers whose properties were isolated (75.6% of ranchers were positive c.f. 38.8%). Ten percent of negative ranchers indicated that they will be positive towards wild dogs when "we form a CNR with neighbours / start ecotourism operations". Attitudes also varied with land use ($p<0.0001$) - cattle ranchers were least positive (16.1% of ranchers were positive), followed by ranchers combining cattle ranching with consumptive use of wildlife (32.0%), ranchers deriving income primarily from consumptive wildlife utilisation (40.4%), and finally by ranchers involved in ecotourism or who did not utilise their property to maximise economic benefits (74.0%). Attitudes were not influenced by ranch size ($p=0.384$).

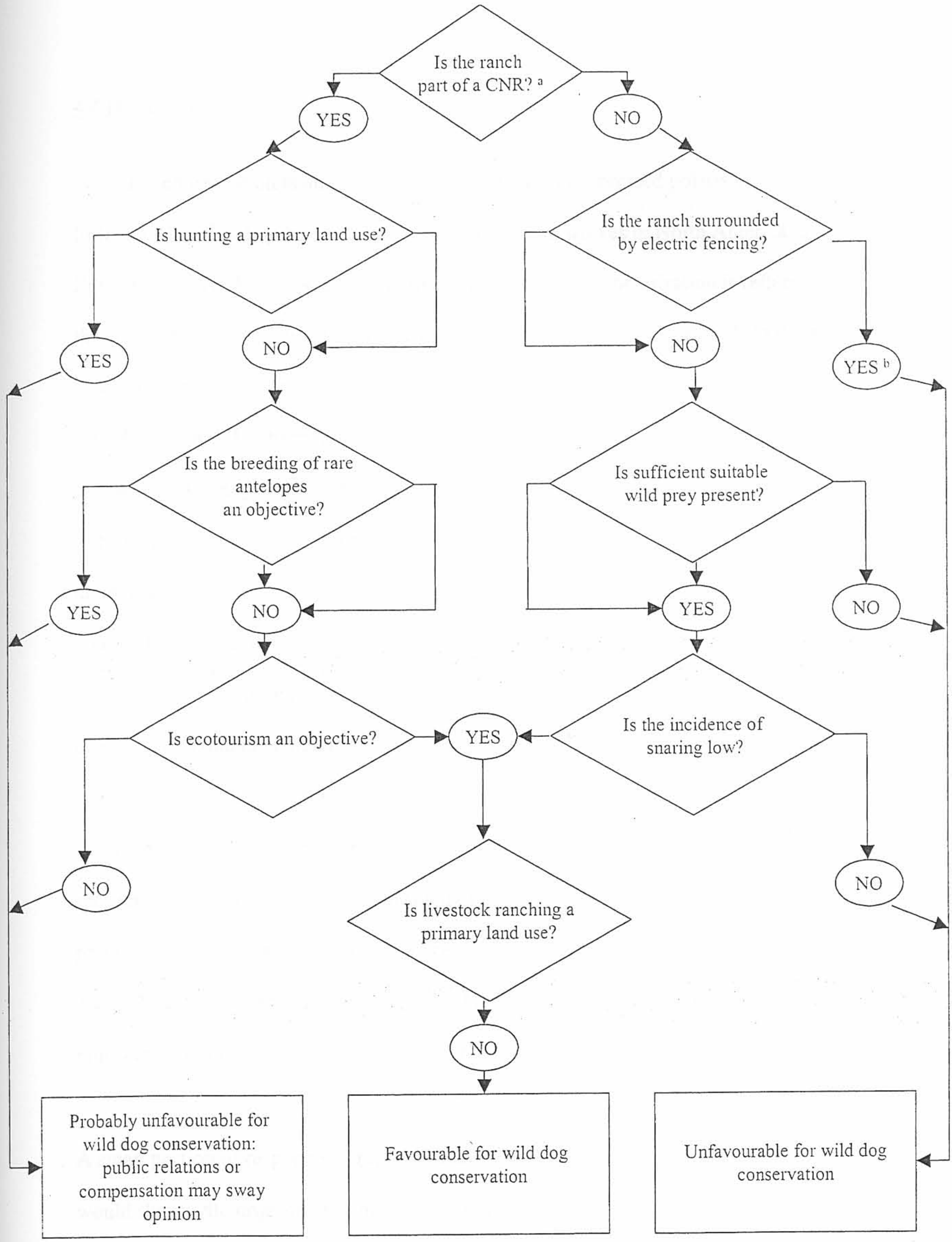

Figure 4.2 General conditions under which wild dog conservation on private land is most likely to succeed

[a] Collaborative nature reserve.

[b] Where fencing prevents wild dogs from accessing a property.

4.4 Discussion

Wild dog conservation is an emotive topic, and ranchers expressed polarised views with little middle ground, in keeping with public opinion on wolves in North America and Europe (Bangs & Fritts 1996; Zimmerman et al. 2001). This polarisation is reflected in the contrast between comments made by positive ranchers, for example: "Wild dogs are endangered and enhance the value of the area"; "We will guard them with our lives"; "Wild dogs appeal to tourists more than lions"; "I love them as animals and they are great for guests", and negative ranchers, for example: "I shoot first and then ask questions"; "The game goes totally wild when wild dogs are on the property"; "It is not just the animals they kill, they also chase game through the fences"; "Wild dogs have no value, and hunting substitutes their ecological role". Wild dogs are the least popular large carnivore species among ranchers, while leopards and black backed jackals are popular due to their economic value through ecotourism and hunting, and low perceived threat, respectively. In line with this, (Bowler 1991) found that wild dogs were the least popular species among Zimbabwean ranchers and leopards the most popular. Southern African ranchers, in contrast to their Kenyan counterparts (Frank & Woodroffe 2001), were not particularly negative towards spotted hyaenas, and many ranchers acknowledged their "ecological role" and their importance in "cleaning the bush of carcasses" (26.9% of ranchers) and their value for ecotourism (19.7%).

A small portion of respondents (3.7% of ranchers) indicated without prompting that they would shoot wild dogs on their property irrespective of the species' legal status, and reported incidents of persecution are common (Chapter 2). Land outside protected areas in South Africa is highly fragmented through the conversion of natural habitat, high human densities and the presence of high-speed roads (Chapter 2). The effect of negative ranchers, is to further fragment natural habitat by creating a mosaic of areas in which wild dogs are protected, and areas in which they are persecuted. By virtue of their obligatorily cooperative breeding system, wild dogs are intolerant of persecution (Courchamp & Macdonald 2001). Although wild dogs continue to survive in ranching areas in several parts of South Africa, they inhabit at most 22.2% of the potentially suitable available habitat (Chapter 2), and it is likely that the minority of negative ranchers seriously limits the survival of wild dogs on private land.

4.4.1 Conditions conducive to conflict between wild dogs and ranchers

The most common reasons provided for negativity towards wild dogs are based upon economic costs associated with their presence, and attitudes are likely to be negative under land use conditions where predation by wild dogs causes economic loss. Cattle ranchers often complained that wild dogs harass and / or kill livestock, whilst many game ranchers complained that wild dogs kill ungulates that could be utilised for hunting or live capture and sale. More ranchers complained about the impact of wild dogs on wild ungulates than about any other carnivore species. Ranchers whose properties were surrounded by perimeter fencing, and not part of CNRs also tended to be negative. Under these conditions, ranchers frequently complained "wild dogs disrupt wildlife during hunting and make it wild", a longstanding complaint held by managers towards wild dogs (Creel & Creel 2002). Under natural conditions, this is not the case, and calm returns rapidly to prey following pursuit by wild dogs (Creel & Creel 2002). However, fencing surrounding small ranches is likely to impede the ability of prey to escape, and as a result prey occurring within fenced areas may be chased for longer (before the dogs give up), or more frequently, both of which may increase prey stress-levels. Many ranchers with fenced properties also complained that wild dogs chase wildlife into game fences, causing damage to, and the loss of wild ungulates through the fencing. Experience from the reintroduction of wild dogs into medium sized reserves (500 km^2 – 650 km^2) suggests that wild dogs use fencing as a tool during hunting (Hofmeyr 1997; van Dyk & Slotow 2003), and this tendency is likely to be exacerbated in small, fenced game ranches.

4.4.2 Strategies to improve rancher's attitudes

The few studies considering the economic costs of large carnivores on ranchland in Africa have suggested that losses due to predation are insignificant relative to other sources of mortality (Mizutani 1993; Rasmussen 1999). Potential economic costs of predation by wild dogs on game ranches in South Africa are high (Chapter 5), however, suggesting that negative attitudes are not entirely without foundation. The negative effect of costs due to wildlife upon the attitudes of local people towards conservation is well established (Infield & Namara 2001; Walpole et al. 2001), and consequently, reducing costs and creating benefits from the conservation of wild dogs represents the most effective way in which attitudes might be improved.

Research into the behavioural ecology of wild dogs under game and livestock ranching conditions is needed to assess economic losses resulting from predation relative to other causes. Rasmussen (1999), for example, found that landowners exaggerated the impact of wild dogs upon cattle and showed that appropriate stock management could greatly reduce losses. Research is also required to assess potential ecological benefits conferred by the presence of wild dogs under game ranching conditions, and it is important to determine the extent to which predation by wild dogs is compensatory, resulting in the death of animals that would have died anyway. Pole et al. (in prep.) for example, showed that wild dogs select for the least fit animals. Wild dogs may also prevent over population of impala *Aepyceros melampus*, which may be detrimental to more sensitive (and more valuable) species such as sable *Hippotragus niger* and roan antelope *Hippotragus equinus*. It is important that these potential benefits are quantified so that landowners consider the positive ecological impact of wild dogs, in addition to economic costs. Research is also required to dispel (or otherwise) the idea that wild dogs disrupt prey populations more than other predators as a result of their cursorial hunting technique. Although this suggestion is largely rejected by scientists (Mills & Nel 1993; Creel & Creel 2002), the prevalence of this belief among ranchers suggests that evidence is required to prove that wild dogs do not have this effect in fenced game ranches.

Fifty six percent (55.5%) of negative ranchers indicated that a way in which to generate sustainable profit from wild dogs would be sufficient incentive for them to want wild dogs on their properties. Wild dogs are popular among tourists (Fanshawe et al. 1991; Davies 1998; Chapter 5) and the financial benefits of wild dog-based ecotourism are predicted to be sufficient to offset the costs of their conservation in game ranching areas under most conditions (Chapter 5). Where feasible, conservation efforts on private land should focus on encouraging and assisting landowners to establish wild dog-based ecotourism ventures. However, although the market for specialised ecotourism is rising with increasing numbers of tourist arrivals to South Africa, tourism is highly susceptible to political instability and it would be unwise to promote wild dog conservation solely on the basis of tourism-related incentives (Sillero-Zubiri & Laurenson 2001). Education programmes, aimed at installing a conservation ethic and reducing misconceptions about wild dogs represent an important additional strategy. For example, 12% of ranchers are under the mistaken impression that wild dogs regularly kill more food than they need to survive. Wild dogs live on a metabolic knife edge as a result of their hunting technique and high daily energy expenditure (Gorman et al. 1998), and as a result, simply cannot afford to waste energy on catching prey other than that required to eat. An education programme bordering Venetia-Limpopo Nature Reserve in northern South Africa yielded assurances that wild dogs leaving the reserve will not be shot (H. Davies pers. comm.).

Finally, encouraging ranchers to form or join CNRs is likely to reduce conflict and create conditions conducive to wild dog conservation. Many of the problems associated with conserving wild dogs on private land are absent in CNRs, due to the absence of internal fencing, the presence of larger prey populations, and economic conditions conducive to ecotourism rather than the consumptive utilisation of wildlife (Barnes & de Jager 1996). Ranchers surveyed within CNRs tended to view wildlife as a communal resource and were less aggrieved by the loss of wild ungulates to predators. Ten percent of ranchers negative towards wild dogs indicated that they will be more positive towards wild dogs when a CNR is formed in their area, or when they start ecotourism operations on their land. CNRs typically have a constitution driving common-decision making concerning wildlife management (Barnes & de Jager 1996; Lambrechts 1996), reducing the likelihood of ranchers persecuting wild dogs. The goal of many CNRs is to reconstruct intact wildlife communities including predators (Lambrechts 1996). Consequently, CNRs provide realistic sites for the reintroduction of wild dogs and / or suitable habitat for natural re-colonisation. The suitability of CNRs for wild dogs (and large predator conservation in general) was reflected in the higher rates of occurrence of wild dogs (and other carnivores) on Zimbabwean ranches (wild dogs frequently sighted on 91.1% of ranches), relative to South African ranches (frequently sighted on 22.7% of ranches). Almost seventy percent (68.8%) of ranches sampled in Zimbabwe were part of CNRs, compared to 28.3% of ranches sampled in South Africa. A graphic example of the suitability of CNRs comes from Save Valley Conservancy in southeastern Zimbabwe, which was re-colonised by wild dogs in the early 1990s, and now has a population estimated at over 100 individuals (A. Pole pers. comm.).

4.4.3 Potential for private land to contribute to wild dog conservation

Over half (52.3%) of ranchers interviewed would like to have wild dogs on their properties. Attitudes are largely positive under conditions in which the costs associated with wild dogs are low, and / or where they are beneficial to economic objectives. Conservation efforts are most likely to succeed where eco-tourism is a primary land use, on properties where economic benefits are not maximised as a management objective, and within CNRs. In keeping with this, 21.6% of ranchers recognised wild dogs as being 'draw cards' for tourism, and several ranchers (13.5%) also recognised the "ecological role" of wild dogs. Furthermore, in addition to recognition of the utilitarian role of wild dogs, 5.8% of ranchers were positive because of their aesthetic appeal. Several ranchers

(7.2% of ranchers) acknowledged the conservation significance of wild dogs on their land, and some expressed a keenness to assist actively in conservation efforts in their area through monitoring, and in helping prevent wild dog persecution by other ranchers.

Conservationists and researchers tend to view ranchers as an obstacle to wild dog conservation (Fanshawe et al. 1991; Rasmussen 1999). Although much needs to be done to improve attitudes and reduce persecution, this view may be unduly pessimistic. In Africa, the last two decades has seen an increase in community based wildlife management schemes in communally owned areas neighbouring protected areas, and game ranching on private land, and there is an increasing awareness that people are part of the solution to conservation problems outside protected areas (van der Waal & Dekker 2000; du Toit 2002). Contrary to previous beliefs, my study suggests that ranchers in southern Africa have the potential to act as important facilitators in the conservation of wild dogs. At present, an estimated 76 wild dogs, in 17 packs and dispersing groups occur outside protected areas in South Africa, primarily on private game ranchland (Chapter 2). There are an estimated 4,000 game ranches in South Africa, covering 80,000 km^2, compared to the 28,000 km^2 under the control of South African National Parks Board (Hearne & Mackenzie 2000). Assuming the same proportion of ranchers are positive towards wild dogs countrywide, as was recorded in the South African sample sites (52.3% of ranchers), then up to 41,840 km^2 of game ranch land is potentially suitable for wild dog conservation. Assuming that wild dogs would reach the lowest density (16.7 dogs / 1000 km^2) observed in Kruger (Maddock & Mills 1994), given adequate protection, this area is potentially capable of supporting ~ 699 adults, or ~ 70 packs. Although some of this land area is likely to be unsuitable for wild dog conservation because of habitat fragmentation or unsuitable land uses, there is nonetheless significant potential for range expansion of wild dogs on private land in South Africa.

The need for increasing the numbers and range of wild dogs in South Africa is evident; the current South African wild dog population numbers as few as 270 individuals (Chapter 2), which yields a demographically effective population size of 213 (Creel & Creel 2002), highly vulnerable to environmental and demographic stochasticity. Although significant potential exists for expanding the range of wild dogs in the future following establishment of the proposed transfrontier parks (www.peaceparks.org), at present, a shortage of suitable state-owned conservation areas dictates that increasing the South African wild dog population will be partially dependent upon reintroduction into private nature reserves and the conservation of naturally occurring wild dogs *in situ* on private land. For this to succeed, conservation efforts aimed at capitalising on existing goodwill, and improving attitudes among negative ranchers through the reduction of costs and maximisation of economic benefits, are required.

In conclusion, identifying the basis of conflict between humans and predators through social outreach represents a vital tool in the conservation of large carnivores, and is an increasingly common feature of species recovery programmes (Messmer et al. 1999). The findings of my study suggest that although much needs to be done to improve the attitudes of a minority of ranchers who persecute wild dogs, significant potential exists for conserving wild dogs on private land in South Africa. To date, this potential has been largely unrecognised. In keeping with findings relating to human-elephant *Loxodonta africana* conflict (Messmer et al. 1999), no single measure can be universally applied to reduce conflict, and judiciously tailored conservation plans for specific areas are required to promote coexistence across a broad spectrum of ranching conditions. Conservation efforts in areas dominated by ecotourism should be based primarily upon the provision of technical assistance for the establishment of wild dog-based ecotourism. Efforts in areas dominated by the consumptive utilisation of wildlife and cattle ranching should be based primarily upon education programmes and the reduction of economic costs associated with wild dogs (for example by encouraging the formation of CNRs, or through appropriate livestock husbandry techniques).

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CHAPTER 5

The potential contribution of ecotourism to wild dog *Lycaon pictus* conservation

5.1 Introduction

Three related economic reasons play an important role in perpetuating habitat conversion and species loss: a) the lack of information on the true value of ecosystem goods and services; b) the failure of markets to capture the benefits provided to human welfare by nature; and c) the promotion of environmentally harmful agricultural activities by governments through perverse subsidies (Constanza et al. 1998; James et al. 1999; Balmford et al. 2002). For example, unprofitable sheep farming in the Swiss Alps is heavily subsidised, creating conditions conducive to conflict with large carnivores (Breitenmoser 1998). The life history of carnivores renders them particularly susceptible to human activities, and the protection of economic interests has been the basis for state-sponsored persecution directed at large predators worldwide (Fanshawe et al. 1991; Berg 2001). It is imperative that conservation strategies are designed to reduce costs incurred by local communities, and to promote exploitation of the use values of large carnivores which can be captured by conventional markets to create financial incentives for conservation. The economic impact of wildlife has a strong influence upon the attitudes of people towards conservation - costs typically result in negative attitudes, whereas benefits generally result in positive attitudes (Infield 1988; Walpole 2001). A major challenge, however, lies in providing benefits to those bearing the costs (Wells 1992) in the form of damage caused by wildlife and / or opportunity costs resulting from resources tied up in conservation (Moran 1994). The conservation of carnivores on private land provides a clear example of the mismatch between those wishing to conserve, and those facing the costs - in many cases, landowners are forced to bear costs associated with carnivores, while society at large benefits from their 'existence value' (Rasker & Hackman 1995). The regulatory approach adopted for many endangered carnivores, based upon the enforcement of legal protection has exacerbated this, and made such species a perceived liability to landowners (Main et al. 1999). Ranchers in southern Africa, for example, complain that although they are forced to bear substantial losses as a result of predation by wild dogs, legal protection prevents the derivation of economic returns from them through hunting (P. Lindsey unpubl. data).

Attempts to promote carnivore conservation outside protected areas in North America and parts of Europe have focused upon the compensation of farmers affected by livestock losses (Mech 1998; Zimmerman et al. 2001), while in Africa, there has been a proliferation of schemes aimed at sharing benefits from the consumptive utilisation of wildlife with affected communities (Hackel 1999). There has also been an increasing interest in the potential for ecotourism to offset the costs associated with the conservation of protected areas and threatened species (Engelbrecht & van der Walt 1993; Norton-Griffiths 1995; Barnes 1998), with particular focus upon primates (Gossling 1999) and elephants *Loxodonta africana* (Brown & Henry 1989). Ecotourism is a rapidly growing industry, and developing nations are increasingly popular destinations (Goodwin 1996; Gossling 1999). Furthermore, a number of studies have revealed significant consumer surpluses among visitors to African protected areas (Moran 1994; Brown et al. 1995;

Barnes & de Jager 1996). The capture and distribution of this consumer surplus to people adversely affected by wildlife has the potential to improve attitudes towards conservation (Infield 1988). By virtue of their popularity with tourists, large carnivores represent ideal targets for conservation strategies based upon the promotion of co-existence with humans using ecotourism-related benefits. In this study, the potential for ecotourism benefits to promote the conservation of an endangered carnivore is investigated, using wild dogs as a model species. Wild dogs are a charismatic species popular with tourists (Davies 1998), and yet remain widely persecuted as a result of perceived or real economic costs associated with their presence (Chapter 4). An assessment is made of the extent to which ecotourism benefits can offset the costs, and provide incentives for the conservation of wild dogs under various scenarios. Here only the ecotourism-related economic benefits associated with wild dogs are addressed. Ecological benefits conferred by the presence of wild dogs are more complicated and require more data.

5.1.1 Current status of wild dogs in South Africa

In South Africa wild dogs occur in three distributions, a single viable population occurring in Kruger National Park (henceforth referred to as "Kruger" ~ 177 – 434 individuals, Maddock & Mills 1994; Davies 2000), a meta-population (~ 54 dogs pre-denning season 2002), and a small population occurring primarily on privately owned ranchland (~ 76 dogs, Chapter 2). The meta-population (n = 6 sub-populations) is the product of a plan to reintroduce wild dogs into a series of isolated reserves linked by management, conceived in recognition of the risks associated with having a single viable population (Mills et al. 1998). Although this strategy has resulted in some success, it is presently heavily dependent upon donor funding and subsidies from state-owned parks (Chapter 6). In addition to suitable state-owned reserves of sufficient size, private nature reserves represent potentially suitable sites for the expansion of the metapopulation. However reintroduction into private nature reserves may depend upon ecotourism benefits being able to offset the costs. Wild dogs occurring on ranchland are few in number and heavily persecuted (Chapter 2). Although an increase in game ranching and an increasing number of collaborative game reserves (Lambrechts 1996) has improved the habitat suitability of ranchland in many areas, an improvement in wild dog status is unlikely to occur in the absence of economic incentives for landowners.

5.2 Methods

Costs, and potential tourism benefits were estimated for combined wild dog conservation / ecotourism operations in three scenarios, using a pack as the functionally minimum demographic unit. 1) Within a viable population (Kruger). 2) Through reintroduction into a private nature reserve. Here, it was assumed that wild dogs are absent prior to reintroduction, and are prevented from leaving the reserve post release by the presence of perimeter fencing. Within this scenario there is a gradation of private nature reserve types from 'ecotourism reserves' in which predation by wild dogs is perceived to result in no cost, to reserves in which land use involves some consumptive utilisation of wildlife, and predation is perceived to result in direct cost to the reserve owner. 3) On livestock / game ranchland. Although some of the ranches in such an area may in fact be private nature reserves, this scenario was distinguished from the reintroduction scenario by the fact that wild dogs occur naturally, without requiring reintroduction, and that wild dogs are able to pass between ranches, due to the absence of predator proof fencing between properties. Countless potential scenarios make it difficult to estimate costs, and consequently estimates were made only for the most likely circumstances. The sum of the discounted costs, and potential tourism benefits were calculated for each scenario in perpetuity, enabling a comparison of all current and future costs associated with conserving wild dogs under different scenarios, in present terms.

5.2.1 Costs per pack within a viable population (Kruger)

The costs of conserving a viable population of wild dogs are impossible to distinguish from the costs of conserving a large protected area in general, and consequently only costs related directly to the conservation of wild dogs were considered. Expenditure records were derived from Kruger staff during direct interviews and an estimate made of annual expenditure. From this, an estimate of expenditure per pack was derived, based upon the average number of packs counted in the last three photographic censuses in Kruger (28 packs: Maddock & Mills 1994; Wilkinson 1995, Davies 2000). Kruger covers a large area (~ 20,000 km^2) and losses of ungulates to predation by wild dogs were assumed to carry no economic cost. Costs were predicted to be limited to veterinary input for the removal of snares, and a five yearly photographic census, during which every individual in the population is identified. Although these activities are not vital for the maintenance of a viable population, the removal of snares reduces human induced mortality, and the five yearly census is conducted to monitor population fluctuations and permit reaction in the event of unusual mortality sources such as disease outbreaks.

5.2.2 Costs per pack of reintroducing and maintaining wild dogs on a private nature reserve

Cost estimates were made for each step of the reintroduction of a pack of wild dogs into a reserve of 360 km^2, equal to the smallest home range size observed in the Kruger (Mills & Gorman 1997), and their post-release maintenance. These costs include: the upgrading of perimeter fencing and pre-release holding facilities (bomas); the capture and transport of wild dogs for reintroduction; vaccination; holding in bomas; monitoring; the genetic and demographic maintenance of the pack post-release and in some cases; the replacement of wild dog prey. Cost estimates were based upon the reintroduction of an average sized newly formed pack (~ 6 dogs, McNutt 1996), assuming that the pack increases to the average pack size observed in Kruger (10 dogs; the mean number of dogs over nine months in age, plus the number of pups divided by two, Mills & Gorman 1997) and is managed to maintain numbers close to this level. Although wild dog numbers are likely to fluctuate somewhat despite management, an average of ten dogs was used to estimate costs. Quotes for services and goods required for the reintroduction and maintenance of a pack were obtained from recognised state agencies, or from three private companies (where possible) and the intermediate quote used (Appendix F).

Cost estimates for perimeter fencing and boma facilities were based on two scenarios - where adequate fencing and boma facilities already exist, and where existing fencing and boma facilities require upgrading to 'wild dog specifications' (Hofmeyr 2000). The first scenario is probably more likely as the reintroduction of lions *Panthera leo* is a priority for attracting foreign visitors to a reserve (Vorhies & Vorhies 1993) and thus adequate boma facilities and fencing are likely to be present prior to the reintroduction of wild dogs. It was assumed that the shape of the theoretical 360 km^2 reserve approximates a square.

Wild dogs should be kept in the boma for a period prior to release, and a captive period of six weeks, and a feeding regime of 2.5 kg meat / dog / day (van Heerden 1992), or two impala *Aepyceros melampus* carcasses per week was used for cost estimates. Mileage, labour and ammunition requirements during hunting were based upon records from reintroductions at Madikwe Game Reserve, Pilanesberg National Park and Venetia Limpopo Nature Reserve.

Suitable captive (Frantzen et al. 2001) and free ranging wild dogs are readily available and it was assumed that founders could be acquired without charge. The costs of capturing free ranging wild dogs for reintroduction are difficult to evaluate, and for the purpose of this study they were based upon 3 hr of helicopter flying time (assuming a den location is known), two days labour from a team of one wildlife veterinarian, one skilled foreman, 20 labourers, and the transport of capture equipment, based on the distance between Kruger and the nearest urban centre (130 km). Costs of transporting captured wild dogs were based upon the average distance between previous reintroduction sites and Kruger, a likely source for future founders (550 km).

It was assumed that wild dogs should be vaccinated against rabies (Hofmeyr et al. 2000), with a combination drug containing vaccines for rabies and several other diseases. This would be administered through intra-venous injection during the initial capture, and a booster rabies vaccine provided following immobilisation in the boma.

Cost estimates for indemnity cover insurance for protection against liability claims arising from damages to people or property as a result of the reintroduction were based on an annual premium for liability cover of ZAR 5 million = $450,000. Monitoring reintroduced animals post release is a vital part of any reintroduction programme (Stanley-Price 1991) and costs include labour, vehicle use, vehicle depreciation and the purchase of telemetry equipment. It was assumed that three of the founder wild dogs would be radio-collared. Kilometers driven was used as a index of monitoring effort - for the first six months, it was assumed that monitoring is conducted at a rate equal to that at Venetia-Limpopo Nature Reserve in the first year post-release (4,000 km monthly), at half this intensity for the second six months, and thereafter, at quarter this intensity. Similarly, for the first year post-release, it was assumed that 100% of an employees' time is spent upon monitoring the reintroduction, decreasing to 25% thereafter. Following reintroduction, it was assumed that three dogs are immobilised annually to replace radio-collars and add collars to young individuals. With adequate habituation following release, wild dogs can be re-captured from the reintroduction site by darting from a vehicle and the costs will include vehicle usage, veterinary labour, and capture drugs.

Although reintroduction of wild dogs into an enclosed reserve will result in genetic isolation, managed gene flow based upon the natural reproductive lifespan of wild dogs (five years) is sufficient to ensure population persistence (Mills et al. 1998). Cost estimates were arbitrarily based upon the removal of six surplus wild dogs and the addition of six individuals, once every five years.

Predation by wild dogs post-release represents a potential additional cost, the extent depending upon the land use of the reserve. Three cost scenarios were considered: 1) where all prey killed results in direct cost; 2) where half of prey killed results in a cost (given reduced intensity recreational hunting of wild dog prey species); and 3) where predation results in no costs. Hunting wild ungulates for venison (recreational hunting) is the most common form of wildlife utilisation in South Africa (van der Waal & Dekker 2000) and for scenarios 1 and 2, it was assumed that prey would be replaced at mean market recreational hunting values for each species, as expressed by a sample ($n = 15$) of operators. Sub-adult animals were assumed to be two thirds of the recreational hunting value, and the juvenile animals one third.

Adult male wild dogs need to consume 3.04 kg / day (Nagy 2001), and the daily requirements of an average sized individual were estimated, based upon 0.75 of mean adult mass for wild dogs (Coe et al 1976). As a rule, 61 percent of the body mass of ungulates is made up of flesh (Blumenschine & Caro 1986) and based on this, the daily food requirement estimate was adjusted to provide an estimate of the mass of prey killed / dog / day (3.2 kg), approximating to field estimates of 1.8 - 3.5 kg / dog / day (Fuller & Kat 1990; Mills & Biggs 1993; Creel & Creel 1995).

The costs of predation by wild dogs were based upon prey profiles (Table 5.1) observed in southern Kruger (Mills & Gorman 1997) and Hluhluwe-Umfolozi Park (Kruger et al. 1999), which represent likely prey-profiles for wild dogs in two of the areas in which reintroductions are likely to occur - northeastern and eastern South Africa. Nyala *Tragelaphus angasi* form a substantial component of the eastern South African prey profile and are a valuable species (male nyala value is ~ \$690 cf. ~ \$50 for a male impala). The more costly predation scenario presented by an eastern South African prey profile is likely to reflect situations where wild dogs occur on land in which the breeding of rare antelopes is a priority. The northeastern South African prey profile is likely to be the most common scenario, because impala and kudu *Tragelaphus strepsiceros* are the most common ungulates in most areas in which reintroductions are likely.

Data on the sex and age breakdowns of prey species were unavailable for the Kruger prey-profile and these breakdowns were extrapolated from data collected in similar habitat in southeastern Zimbabwe, adjacent to Kruger (Pole 1999). Prey animals of adult body mass were classified as adult, animals under one year of age were classified as juvenile and animals between these two categories were classified as sub-adults. Mean body-mass for each age class of each species was derived from literature (Simpson 1966; Simpson 1973; Howells & Hanks 1975; Anderson 1978; Anderson 1986; Pole 1999). For species where these data were not available, differences in body mass between age

Table 5.1 Percent biomass made up by each prey species, sex and age class in two wild dog prey-profiles

Area	% of total	% adult male	% adult female	% sub adult	% juvenile
Eastern South Africa [a]					
Grey duiker *Sylvicapra grimmia*	0.1	0.1	0.05	0	0
Impala *Aepyceros melampus*	16.2	4.2	5.6	1.2	5.2
Kudu *Tragelaphus strepsiceros*	0.7	0.1	0.1	0.3	0.3
Nyala *Tragelaphus angasi*	76.1	31.8	29.8	7.8	6.7
Red duiker *Cephalophus natalensis*	0.3	0.2	0.15	0	0
Reedbuck *Redunca arundinum*	0.9	0.3	0.3	0.2	0.1
Waterbuck *Kobus ellipsiprymnus*	2.2	0.2	0.3	0.9	0.8
Wildebeest *Connochaetes taurinus*	3.5	0.3	0.5	1.4	1.3
Total	100				
Northeastern South Africa [b]					
Bushbuck *Tragelaphus scriptus*	2.0	0.5	0.6	0.3	0.2
Grey duiker *Sylvicapra grimmia*	4.4	2.2	2.2	0	0
Impala *Aepyceros melampus*	81.0	18.6	25.7	13.0	8.6
Kudu *Tragelaphus strepsiceros*	8.1	0.3	0.6	2.2	4.1
Reedbuck *Redunca arundinum*	2.0	0.5	0.6	0.3	0.2
Steenbok *Raphicerus campestris*	2.5	1.3	1.3	0	0
Total	100				

[a] Kruger, Lawes & Maddock (1999).
[b] Mills & Gorman (1997).

categories were estimated from the species closest in size for which data do exist. Estimates were made of the number of individuals of each sex / age category, and species killed annually by 10 wild dogs, under both prey-profiles, and cost estimates were calculated for each.

5.2.3 Costs of conserving a wild dog pack on ranchland

The costs of conserving wild dogs on ranchland were estimated for a pack of 10 dogs. It was assumed that ranchers would adopt a minimum approach to a wild dog ecotourism / conservation programme, to maximise the extent to which ecotourism revenues can offset the costs resulting from the presence of wild dogs. In this scenario, costs were limited to those resulting from predation by wild dogs, assuming that no collaring or monitoring would occur. Wild dogs on private land are likely to come into contact with livestock and each prey-profile is assumed to include the same proportion of cattle (32.2%) observed in the sole published study of wild dogs in a ranching area (Rasmussen 1999). Cattle prices were obtained from the First National Bank Department of Agricultural Information.

5.2.4 Potential revenue from wild dog based ecotourism

Contingent valuation methods involve the construction of a contingent market, in which respondents express their willingness to pay (WTP) for quantitative and / or qualitative changes in a good (Mitchell & Carson 1989). Typically, contingent valuation methods are used to measure the total economic value of non-market goods. The types of values associated with non-market goods include non-use values such as option, existence and bequest values, in addition to use values such as tourism or hunting (Loomis & White 1996; White et al. 2001). In my study, however, contingent valuation was used to measure the value of a market good - opportunities to view wild dogs with puppies at a den site. In this case, total economic value is comprised of use-value, with little or no non-use value. Many of the problems associated with the conventional usage of contingent valuation associated with measuring non-use values for non-market goods (Mitchell & Carson 1989), disappear when measuring use values. Target respondents are easily identified, removing the risk of 'population choice bias', the 'goods' in question are easily explained, and respondents are experienced in making similar transactions, removing the risk of 'scenario misspecification bias' (Mitchell & Carson 1989). Finally, the risk of 'embedding', whereby WTP bids are influenced by the moral satisfaction of paying for an environmental good, with little sensitivity to the scope of the good (Carson 2000) is lower in studies of direct use value (Navrud & Mungatana 1994), and absent in my study because respondents were clearly offered no more than a simple tourist experience.

I used a questionnaire (Appendix G) with an open-ended question designed to determine the willingness of visitors to protected areas to pay for wild dog-viewing opportunities. Open-ended questions tend to yield conservative WTP estimates, and remove the potential for 'starting-point bias' associated with close-ended formats (Navrud & Mungatana 1994). From the WTP bids obtained, I calculated the mean willingness of tourists interested in seeing wild dogs to pay by removing zero bids, and estimated potential annual revenues from wild dog-based ecotourism. It was assumed that one viewing trip could be made to a den in the morning, and one in the late afternoon, with a safari vehicle capable of holding nine guests, over a period of three months during denning season. Four scenarios were presented for each WTP estimate, based upon 100%, 75%, 50% and 25% bookings. A 75% booking rate was assumed to represent an approximation of the likely revenue from a successful wild dog-ecotourism operation and was used to discuss the potential for benefits to offset the costs of conservation efforts. The costs associated with running an ecotourism operation of this nature, including the salary of a guide, vehicle-running costs (based on a round trip of 40 km to the den), and the costs of monthly advertising in a national wildlife magazine were subtracted from the revenue estimates.

Sampling was done at two public reserves, Pilanesberg National Park (PNP) and Kruger, to obtain WTP estimates from low to medium budget tourists, and at two private nature reserves, Djuma Game Reserve (DGR) and Ngala Game Reserve (NGR), to obtain WTP bids from "up market" tourists. Pre-testing was done at PNP to highlight problems with draft questionnaires. The following question was used to determine tourists' WTP to see wild dogs (Appendix G):

> *Although wild dogs are present at Pilanesberg / Kruger / Ngala / Djuma, you have a less than 5% / 10% / 20% / 20% chance of seeing them during your stay. For three months in winter, wild dogs remain in the vicinity of the den in which they have pups. During this time, the location of wild dogs is very predictable and guided trips to see them would be almost guaranteed sightings. How much would you pay per person, to go on an optional, small (six persons, max) guided tour to a den, to improve your chances of seeing wild dogs to more than 90%?*

Direct interview surveys were conducted at PNP and Kruger. At PNP, sampling was stratified between camps and restaurants to obtain coverage of all market segments. In Kruger, sampling was done at Skukuza, the administrative park 'capital', and at neighbouring picnic sites, providing coverage of visitors based at camps throughout the park. Refusal rate at both parks was <5%. Interviewing was not permitted at DGR and NGR, and questionnaires were distributed to all guests on arrival to complete during their stay. A total of 596 completed questionnaires were obtained from the four sample sites.

5.2.5 Present values (PVs) of revenue from wild dog ecotourism operations

All cost and benefit estimates were converted into US$, using the mean US$ / ZAR exchange rate for the first sixth months of 2002 ($1 = R10.99). Future costs and benefits were discounted (using average long-term South African Government Bond rates for the first six months of 2002) to yield PVs. In my calculation of the costs and benefits incurred under the three scenarios, it was assumed that costs and benefits would stabilise quickly. In reality however, due to a variety of potentially unforeseen factors, it may take some time for costs and benefits to stabilise. Consequently, the calculation of costs and benefits into PVs is expressed in a general form, assuming that it would take five years for costs and benefits to stabilise:

$$TC = SSC + \frac{C_1}{(1+r)} + \frac{C_2}{(1+r)^2} + \frac{C_3}{(1+r)^3} + \frac{C_4}{(1+r)^4} + \frac{C_5}{(1+r)^5} + \frac{\frac{C_5 + \frac{\text{Five-year cost}}{5}}{r}}{(1+r)^5}$$

Where *TC* (total costs) represents the PV of the costs of conserving a pack of wild dogs in perpetuity. The costs and benefits of conserving wild dogs under the three scenarios are predicted to occur over different time scales (for example, on ranchland annual costs are predicted to be constant, whereas within a viable population, significant costs are incurred every five years) and the calculation of PVs of perpetual costs enabled comparison of all costs from each scenario in present terms. *SSC* is the sum of the start up costs of a conservation programme, and r is the discount rate. C_1 - C_5 represent the annual maintenance costs under each scenario over five years, and the penultimate term, where C_5 is located accounts for the continuing costs in perpetuity, assuming that the annual costs in perpetuity will be equal to C_5. The last term in the equation accounts for costs occurring on a five yearly basis (such as the photographic census in Kruger), assuming for tractability that one-fifth of the five-yearly cost occurs each year.

Tourism revenue benefits (TB) were converted into PVs using the following equation:

$$TB = \frac{B_1}{(1+r)} + \frac{B_2}{(1+r)^2} + \frac{B_3}{(1+r)^3} + \frac{B_4}{(1+r)^4} + \frac{B_5(1+\frac{1}{r})}{(1+r)^5}$$

Where, *TB* represents the ecotourism benefits of conserving a pack of wild dogs in perpetuity and B_1 - B_5 represent the annual benefits under each scenario over five years. The term where B_5 is located accounts for the continuing benefits in perpetuity, assuming that benefits in perpetuity will be equal to B_5. The net present value (NPV) of each conservation option was then calculated by subtracting TB from TC: NPV = TB – TC.

5.3 Results

5.3.1 Costs per pack within a viable population (Kruger)

Annual costs associated with maintaining a viable population are predicted to be low, at $76 (Table 5.2). Additional costs of $8,180 are incurred every five years as a result of the photographic census. The present value of the costs of conserving wild dogs in a viable population in perpetuity is estimated at $51,812 for the entire population, or $1,850 per pack (Table 5.3).

5.3.2 Costs per pack of reintroducing and maintaining a wild dog pack on a private nature reserve

Estimated costs of the initial reintroduction are 96.7% greater if upgrading of fencing and boma facilities is necessary ($128,924 compared to $4,265,Table 5.4). Predation by wild dogs potentially causes the greatest predicted cost associated with the maintenance of a pack post-release. Predicted costs are greatly affected both by the prey-profile and by the proportion of the prey that results in costs. Given the worst case predation-cost scenario, whereby all prey killed in an eastern South Africa prey-profile result in cost, the annual costs of maintaining a pack in most years are $85,803, compared to $7,525, given the best case predation scenario, whereby predation results in no cost (Table 5.5).

The PV of costs associated with reintroducing and maintaining a pack of wild dogs in a private nature reserve vary greatly, depending primarily upon fencing requirements,

Table 5.2 The costs in 2002 US$ of conserving a viable population of wild dogs (ZAR in parentheses)

Source of cost	Annual costs [a]	Additional five yearly costs [a]
Snare Removal	76 (830)	-
Photo census	-	
Prizes [b]	-	773 (8,500)
Vehicle use	-	3,949 (43,400)
Computer equipment	-	1,274 (14,000)
Researcher salary	-	2,184 (24,000)
Total	76 (830)	8,180 (89,900)

[a] Costs for the whole population – costs per pack can be obtained by dividing these figures by 28, the average number of packs in Kruger.

[b] Offered as incentives to encourage tourists to assist with the photographic survey.

Table 5.3 Present values of the US$ costs of conserving a pack of wild dogs in perpetuity under three scenarios (ZAR in parentheses)

Pack scenario	PV of costs per pack in perpetuity	
Viable population	1,850 (20,332)	
Private reserve	Fence upgrade	No fence upgrade
ESA prey-profile		
All prey compensated	856,928 (9,417,639)	732,269 (8,047,636)
Half prey compensated	530,224 (5,827,162)	405,565 (4,457,159)
NESA prey-profile [a]		
All prey compensated	317,559 (3,489,973)	192,900 (2,119,971)
Half prey compensated	260,539 (2,863,328)	135,881 (1,493,332)
Zero predation costs	203,520 (2,236,685)	78,861 (866,682)
Ranchland		
ESA prey-profile		
All prey compensated	460,136 (5,056,895)	
Half prey compensated	230,068 (2,528,447)	
NESA prey-profile [b]		
All prey compensated	94,934 (1,043,325)	
Half prey compensated	47,467 (521,662)	

[a] Most likely scenarios for wild dog reintroductions.
[b] Most likely scenario for wild dogs on ranchland.

Table 5.4 Estimated 2002 US$ costs of the initial reintroduction of a pack of wild dogs into a private reserve (ZAR in parentheses)

Item	Source [a]	Estimated cost
Perimeter fence upgrade	1	121,929 (1,340,000)
'Soft release' methods		
Boma upgrade	2	2,730 (30,000)
Hunting to feed wild dogs in captivity	3,4,5,6,7	1,164 (12,792)
Capture of founder wild dogs		
Capture	8,9,10,11	1,980 (21,760)
Transport of wild dogs	10	494 (5,429)
Vaccination	9,10	627 (6,890)
Total without fence and boma upgrade		**4,265 (46,872)**
Total with fence and boma upgrade		**128,924 (1,416,875)**

[a] Sources of quotes are listed in Appendix F.

Table 5.5 Estimated 2002 US$ costs of maintaining a pack of wild dogs in a private reserve (ZAR in parentheses)

Item	Source [a]	1st year costs	Most years	Additional five yearly costs
Insurance	12	1,365 (15,000)	1,365 (15,000)	
Monitoring				
Vehicle use	3, 13	9,001 (98,921)	3,694 (40,597)	
Telemetry equipment	14, 15, 16	1,619 (17,791)	721 (7,922)	
Full time employee	17	4,648 (51,085)	1,162 (12,771)	
Capture to attach collars				
Veterinary expertise	9		280 (3,080)	
Capture drugs	10		303 (3,333)	
Removal of six wild dogs				
Veterinary expertise	9			280 (3,080)
Capture drugs	10			303 (3,333)
Transport of wild dogs	8			256 (2,814)
Addition of six wild dogs				
Vaccination	9,10			573 (6,297)
Radio collars	14			721 (7,922)
Pre-release holding	3,4,5,6,7			1,164 (12,789)
Predation				
ESA prey-profile				
All prey compensated		78,278 (860,279)	78,278 (860,275)	
Half prey compensated		39,139 (430,139)	39,139 (430,139)	
NESA prey-profile				
All prey compensated		13,662 (150,154)	13,662 (150,154)	
Half prey compensated		6,831 (75,072)	6,831 (75,072)	

[a] Sources of quotes are listed in Appendix F.

observed prey-profile and the proportion of prey that is replaced (Table 5.3). Maximum and minimum estimated PVs of the costs of the reintroduction and maintenance of wild dogs in a private nature reserve were 463 and 43 times greater than the PV of the costs of conserving a pack within in a viable population, respectively.

5.3.3 Costs of conserving a wild dog pack on ranchland

Given an eastern South African prey-profile, annual costs of conserving wild dogs on ranchland were predicted to be $27,562 if one assumes that half of prey killed results in cost, or $55,124, assuming that all prey killed results in cost. Given a northeastern prey-profile, annual costs are estimated to be 79.4% lower, at $5,687 - $11,374. Maximum and minimum estimated PVs are 249 and 26 times greater than the PV of the costs of conserving a pack within a viable population respectively (Table 5.3). The PVs of the costs of conserving dogs on ranchland are lower than the PVs of reintroducing dogs into private nature reserves given equivalent predation-cost scenarios.

5.3.4 Potential revenue from wild dog based ecotourism

Mean WTP was highest at NGR ($53 / per person / trip), followed by DGR ($45 / per person / trip), PNP ($12 / per person / trip) and Kruger ($11 / per person / trip). The annual costs of running wild dog based ecotourism were estimated to be $3,246, given costs of $590 for advertising, $751 for the salary of a guide, and $1,905 for vehicle use and depreciation over the three month denning period (see items 3, 13, 18 and 19 in Appendix F for the sources of cost estimates). Under the Kruger estimate of WTP, estimated annual income (with the costs of the ecotourism operation subtracted) ranged from $1,540 (25% bookings) - $15,895 (100% bookings), compared to $1,878 (25%

bookings) - \$17,287 (100% bookings) under the PNP estimate of WTP, \$15,435 (25% bookings) - \$71,478 (100% bookings) under the DGR estimate of WTP, and \$19,003 (25% bookings) - \$85,748 (100% bookings) under the NGR estimate of WTP (Figure 5.1).

5.3.5 NPV of a combined conservation-ecotourism programme for a pack of wild dogs under each scenario in perpetuity

The predicted NPV of conserving a pack within a viable population in perpetuity was +\$90,887 (Table 5.6). Predicted NPV of conserving a pack of reintroduced wild dogs in perpetuity varied greatly from -\$764,191 to +\$451,184, in relation to the need for perimeter fencing, the prey-profile and the proportion of prey that must be replaced. Predicted NPVs were generally positive given high estimates of tourist WTP, and generally negative given low estimates of tourist WTP. The NPV of a pack of wild dogs on ranchland was predicted to be positive under most scenarios (+\$45,270 to +\$506,311). Given low estimates of tourist WTP and high predation costs, however, predicted NPV was negative (-\$137,331 to -\$367,399).

The NPVs presented are the present value of all present and future costs, and not an amount that a ranching community would pay or receive at any given time. Given the mean ranch size in the areas where wild dogs occur on private land in South Africa (3,047 ha ± 236 S.E., Chapter 4), and the mean home range of wild dogs in Kruger (55,300 ha, Fuller et al. 1992), a pack would traverse ~ 18 ranches. If costs and benefits were distributed evenly across ranchers, assuming that predation results in costs equal to

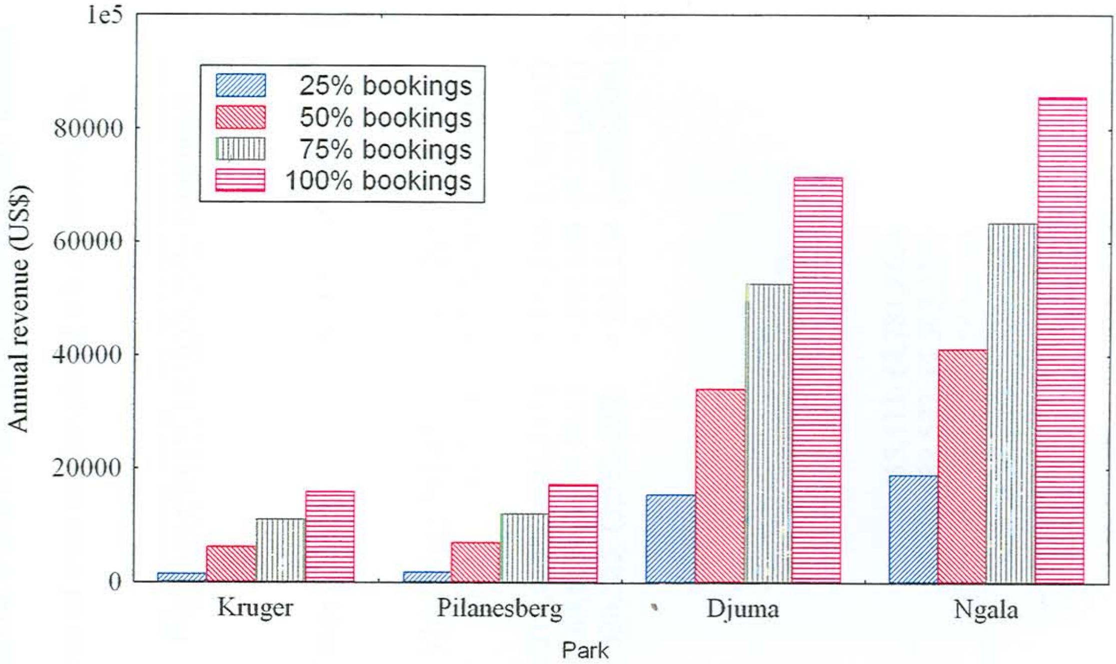

Figure 5.1 Potential annual revenue from wild dog ecotourism under willingness to pay estimates from Kruger, Pilanesberg, Djuma and Ngala

Table 5.6 The predicted NPV in 2002 US$ of conserving a wild dog pack in perpetuity, within a viable population (Kruger), through reintroduction into a private reserve, and *in situ* on ranchland, under various scenarios of costs and benefits (ZAR in parentheses)

Scenario	Low benefits (WTP=$11, 75% bookings)		High benefits (WTP=$53, 75% bookings)	
Viable population	+90,887 (998,848)			
Private reserve	Fence upgrade	No fence upgrade	Fence upgrade	No fence upgrade
ESA prey-profile				
All prey compensated	- 764,191 (8,398,458)	- 639,532 (7,028,458)	- 326,884 (3,592,450)	- 202,225 (2,222,450)
Half prey compensated	- 437,441 (4,807,487)	- 312,828 (3,437,979)	- 179 (1,971)	+ 124,480 (1,368,029)
NESA prey-profile [a]				
All prey compensated	- 224,822 (2,470,795)	- 100,163 (1,100,795)	+ 212,485 (2,335,213)	+ 337,144 (3,705,213)
Half prey compensated	- 167,802 (1,844,147)	- 43,144 (474,147)	+ 269,505 (2,961,861)	+ 394,164 (4,331,861)
Zero predation costs	- 110,782 (1,217,500)	+ 13,876 (152,501)	+ 326,525 (3,588,508)	+ 451,184 (4,958,508)
Ranchland				
ESA prey-profile				
All prey compensated	- 367,399 (4,037,711)		+ 69,909 (768,297)	
Half prey compensated	- 137,331 (1,509,265)		+ 299,948 (3,296,433)	
NESA prey-profile [b]				
All prey compensated	- 2,196 (24,138)		+ 435,111 (4,781,860)	
Half prey compensated	+ 45,270 (497,521)		+ 482,577 (5,303,529)	
Zero predation costs	+ 92,737 (1,019,181)		+ 506,311 (5,564,354)	

[a] Most likely cost scenarios for wild dog reintroductions.
[b] Most likely scenario for wild dogs on ranchland.

half of the prey killed by wild dogs, the annual net economic impact of conserving a pack on ranchland per rancher would be; -$914 given an eastern South Africa prey-profile and the Kruger estimate of WTP; +$1,997 given an eastern South Africa prey-profile and the Ngala estimate of WTP; +$301 given a northeastern prey-profile and the Kruger estimate of WTP; and +$3,212 given a northeastern prey-profile and the Ngala estimate of WTP.

5.4 Discussion

During the denning season the location of wild dogs is highly predictable (Fuller et al. 1992), and they are easily habituated, avoiding the need for contentious supplementary feeding programmes necessary for tourism programmes involving some large carnivore species (Walpole 2001). Wild dogs are charismatic and their conservation plight is well publicised, both factors that increase WTP bids in contingent valuation studies (White et al. 2001). Consequently, there is potential to generate substantial revenue from wild dog-based ecotourism. Furthermore, there is some potential to derive revenue from wild dog-based ecotourism outside of the denning season, if the wild dogs are radio collared and sufficiently habituated. However, this potential was not investigated in the present study by virtue of the fact that due to their wide ranging behaviour, one would not be guaranteed to be able to find the dogs, or to gain access to them if they moved into rough terrain, or away from roads.

The degree to which benefits can offset costs, varies greatly both within and between sectors of the South African wild dog population, in keeping with varying potential for benefits to offset the costs of lion conservation between scenarios (Vorhies & Vorhies 1993; Cotterill 1995). Wild dog conservation is potentially extremely costly, and under the most costly scenario, the conservation of a single wild dog reintroduced into a private nature reserve is estimated to cost $8,840 per year, compared to $273 per year for a single wolf *Canis lupus* in Minnesota (Mech 1998). In contrast to North America, wildlife is the property of landowners in southern Africa (Cumming 1991) and predation upon wild ungulates may result in substantial costs if full values are assumed to apply. Minimum cost estimates ($10 / dog / year) for wild dog conservation are, however, far lower than the maximum estimate, and careful consideration of the predicted costs and benefits represents a vital part of the planning of conservation initiatives.

The conservation of wild dogs within Kruger involves the lowest costs per pack, with most costs being indistinguishable from the conservation of the wildlife population of the park as a whole. Given the mean WTP of tourists at Kruger, and 75% bookings, eco-tourism benefits from one pack ($11,110 / year) are predicted to far exceed the average annual costs of conserving the entire Kruger population ($1,712). Tourist volumes in Kruger are high (Engelbrecht & van der Walt 1993) and almost three quarters of guests are willing to pay to see wild dogs. This, coupled with large consumer surpluses among visitors to African protected areas (Moran 1994; Navrud & Mungatana 1994; Barnes & de Jager 1996), suggests that the market in Kruger may be sufficient to support ecotourism on multiple packs, at rates above the mean WTP. This would generate substantially greater revenue than predicted in this chapter, and would potentially be sufficient to provide funding for conservation initiatives elsewhere in the country.

Benefits derived from conservation are typically invested in the conservation of all species in a given area, as for example occurs with community conservation initiatives in Africa (Archabald & Naughton-Treves 2001). In this case, however, it is suggested that economic benefits from wild dog-based ecotourism in Kruger should be invested into the conservation of the same species elsewhere in the country, enabling wild dogs to contribute towards their own conservation. CITES conditions aim to achieve a similar goal for elephant conservation, albeit using benefits derived from consumptive utilisation (Sharp 1997).

The NPV of the reintroduction and maintenance of a pack in a private nature reserve varies from positive, to prohibitively negative, depending primarily upon tourist WTP, the need for perimeter fencing improvements, and the requirement for, and cost of, prey replacement. Subsequently, careful site selection for wild dog reintroduction is vital. Although under most scenarios presented NPVs were negative, there is reason to believe that many private nature reserves will provide conditions conducive to positive NPVs for wild dog reintroduction. First, there are many private nature reserves with existing predator proof fencing in South Africa (Chapter 4). Second, impala and kudu constitute the primary prey species of wild dogs in most parts of southern Africa (Fuller et al. 1992) and so predation costs are likely to be closest to those predicted for the lower cost northeastern prey-profile. Third, the most profitable land use in reserves of a size sufficient for the reintroduction of wild dogs is eco-tourism (Barnes & de Jager 1996). Under these conditions, hunting is largely precluded by the need for habituated animals for game viewing (Barnes & de Jager 1996; Falkena 2000) and the realised costs of predation by wild dogs are likely to be low or zero. Fourth, wild dogs select for individuals in poor condition (Pole 1999; Pole et al. in prep.) and a portion of prey killed by wild dogs is likely to be compensatory, resulting in the death of individuals that would have died anyway. Fifth, at present, intensive post-release monitoring is done to improve understanding of the ecology and behaviour of reintroduced wild dogs. In future, lower intensity monitoring immediately after a reintroduction, followed by infrequent check-ups thereafter is likely to suffice, which would decrease predicted costs substantially. Finally, many private nature reserves have existing ecotourism infrastructure and a ready source of high-paying visitors. Under these conditions, there is scope for wild dog-based ecotourism to offset the costs of wild dog reintroductions, suggesting that the expansion of the meta-population could be promoted by encouraging private nature reserve owners to reintroduce dogs at their own cost, thereby reducing or removing reliance upon donor funding. There is an ongoing proliferation of private nature reserves worldwide (Langholz 1996), and particularly in South Africa (Lambrechts 1996; Chapter 4) and increasing realisation that developing nations can effectively expand formal park systems by utilising these private nature reserves for the conservation of large carnivores, and other species (Langholz et al. 2000).

The logistics involved in conserving naturally occurring wild dogs on ranchland are simpler, and less costly than required for reintroduction programmes, and revenue from ecotourism is predicted to exceed the costs under most scenarios. Negative attitudes among ranchers towards wild dogs are based primarily upon costs associated with predation (Chapter 4) and by offsetting these costs, ecotourism schemes have the potential to improve conditions for wild dog conservation significantly. Where wild dogs occur on ranchland, most ranchers (82.7% of ranchers) derive part or all of their income from consumptive wildlife utilisation or livestock, and ranches are typically small (mean = 3,047 ha) and surrounded by perimeter fencing (Chapter 4). Under these conditions, predation by a single dog is estimated to cost as much as \$32 - \$306 / rancher / year (given 10 dogs in a pack, and assuming the dogs traverse 18 ranches), or \$1 - \$10 / dog / km^2 / year, compared to estimated costs of \$2.6 / lion / km^2 / year incurred by ranchers in Kenya due to predation upon livestock (Frank & Woodroffe 2002). This suggests that negative attitudes among ranchers towards wild dogs are based upon economic fact rather than 'prejudice or ignorance' as often assumed by conservationists (Fanshawe et al 1991; Rasmussen 1999). The focus of conservation efforts involving wild dogs on ranchland should be the establishment of ecotourism schemes to offset the costs borne by ranchers, so as to create conditions more conducive to conservation. The design of appropriate profit sharing schemes for multiple ranchers to ensure that benefits are distributed relative to costs would be a difficult, but vital part of this strategy.

In areas far from traditional tourist routes, however, or where the breeding of rare antelopes is a priority, the establishment of wild dog-based ecotourism may require significant logistic investment, and benefits probably won't match the costs. Under these conditions, combined ecotourism / compensation conservation strategies might be considered, whereby ranchers conduct wild dog-based ecotourism to offset some of the costs associated with wild dog conservation, and NGOs provide compensation for outstanding losses. Once compensation schemes are initiated however, they need to be continued indefinitely and potentially represent 'bottomless pits' for donors. A more sustainable alternative would be to exploit the ecotourism potential of wild dogs as much as possible, and then to invest in educational programmes aimed at highlighting the ecological benefits of wild dogs. Wild dogs select for the least fit individuals in prey populations (Pole et al. in prep.) and ecological benefits conferred by wild dogs may offset their economic impact, by reducing the need for active management in the form of culling, and promoting the genetic 'health' of ungulate populations (Mills 1991). Research is required to quantify the ecological role of wild dogs in ranching conditions to increase the utility of this aspect of behavioural ecology as a tool in educational programmes.

Wild dogs represent an ideal species for investigating the potential role of ecotourism in endangered species conservation. They are persecuted in response to costs associated with their presence, and yet have significant earning potential, by virtue of their popularity with tourists. Nonetheless, the methods in this study are applicable to a wide variety of species. For example, expanding populations of brown bears *Ursus arctos*, lynx *Lynx lynx*, and wolves in Europe, and cougars *Felis concolor* and wolves in North America are increasingly conflicting with private land owners as a result of increasing livestock depredation, and competition with humans for wild prey (Breitenmoser 1998; Mech 1998). In Europe and the USA, tourists do not typically visit private ranches, and as a result, conservationists have largely ignored the potential for exploiting the tourist value of large carnivores. The results of this study suggest that tourists are willing to pay substantial amounts to view large carnivores in their natural habitats, and given the popularity of large carnivores among urban residents in the developed world (Ericsson & Herbelein 2003), the establishment of ecotourism operations on private land in Europe and North America has the potential to create powerful incentives for carnivore conservation. Rapid growth in the ecotourism industry (Gossling 1999) and increasing demand and willingness to pay for ecotourism experiences (Moran 1994) has created unprecedented potential for exploiting tourism values of large carnivores to promote their conservation.

There are, however, limitations to ecotourism as a tool in conservation. It is important to be aware that the benefits may never match the costs (Archabald & Naughton-Treves 2001), and to have alternative strategies to cater for changing economic conditions or political unrest (Wilkie et al. 2001). Ecotourism should not be relied upon in isolation to promote the conservation of endangered species (Sillero-Zubiri & Laurenson 2001). In keeping with this, my study suggests although ecotourism has significant potential under some scenarios, a diverse, adaptive approach is necessary if wild dog conservation is to be considered across land tenure categories.

In conclusion, ecotourism has a potentially significant role in promoting conservation efforts in all three distributions of the South African wild dog population. In Kruger, if wild dog conservation could be isolated from other conservation activities, Kruger would have the potential to act as a source of funding for conservation initiatives on private nature reserves and on ranchland. Ecotourism revenue has the potential to offset the costs of wild dog reintroductions under certain conditions, suggesting that private nature reserve owners might be encouraged to reintroduce wild dogs at their own expense, thus reducing reliance upon donor subsidies for expansion of the meta-population. Ecotourism perhaps has most potential on ranchland, by generating self-sustaining 'compensation' for costs resulting from predation by wild dogs, thus removing the major source of conflict between wild dogs and humans.

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CHAPTER 6

The cost efficiency of wild dog *Lycaon pictus* conservation in South Africa

6.1 Introduction

Cost efficiency in conservation can be gauged in terms of units of an environmental good conserved per unit money spent. Examples of units of 'environmental goods' in this context include, the recovery in numbers of a population of a species, or the land area in km^2 of forest conserved. Pressures on remaining natural habitats are increasing, and the number of threatened species is rising correspondingly (Guikema & Milke 1999). As a result, there is a worsening shortfall between the resources required for conservation, and what is available (Myers et al. 2000). For example, the costs of a globally effective reserve network encompassing terrestrial and marine habitats are estimated at $45 billion per year (Balmford et al. 2002), while current global spending on reserve networks is as little as $1 - 6 billion (James et al. 1999; Balmford et al. 2002). Expenditure on endangered species in the USA is estimated to be 20% of the amount required (Miller et al. 2002) and in developing countries the shortfall is likely to be substantially greater. The gulf between funding requirements and current global spending dictates that cost efficiency should be a primary consideration in conservation prioritisation (Balmford et al. 2003).

Conservation competes for donor funding with other human-centred issues and there has been some political controversy surrounding the amount of money being spent on endangered species conservation (Baker 1999). Consequently, there is an ethical basis for the need for cost efficiency in conservation. In addition, there has been increasing debate surrounding the skewed distribution of donor funds (Czech et al. 1998; Baker 1999), with conservation efforts typically favouring a minority of charismatic species (Restani & Marzluff 2002). Prospects for recovery in populations of threatened species improve with increasing donor funding (Miller et al. 2002; Restani & Marzluff 2002), and cost efficient conservation programme design increases the chances of financial support (Moran et al. 1997). Cost efficiency is of particular importance in developing countries where the shortfall in funding for conservation is largest, and where the potential return from conservation spending is greatest (Balmford et al. 2003).

Cost efficiency has been the focus of a number of studies on conservation planning (Moran et al. 1997; Ando et al. 1998; Balmford et al. 2003), and several studies have reviewed spending in relation to the US Endangered Species Act (Baker 1999; Miller et al. 2002; Restani & Marzluff 2002). Research into the cost efficiency of conservation options involving single species, however, has been less common. Mech (1998) considered the costs associated with two wolf *Canis lupus* conservation options in Minnesota, while Main et al (1998) estimated the costs of two conservation plans for Florida panthers *Felis concolor coryi.* In many developing countries, cost efficiency in conservation planning has received little or no attention, despite a desperate shortage of funds.

In my study, the African wild dog is used as model species to investigate the role of donor funding in the conservation of an endangered carnivore in southern Africa, and to estimate the cost efficiency of current, and potential future conservation strategies.

6.1.1 Current conservation efforts involving wild dogs in South Africa

Despite an historic range comprising almost all of South Africa (Skinner & Smithers 1990), wild dogs are currently limited to a single viable population (177 – 434 individuals in 21 – 32 packs, Maddock & Mills 1994; Davies 2000), occurring in the Kruger National Park (henceforth referred to as "Kruger"). Most of South Africa (~ 70%) is comprised of private land, and state-owned protected areas comprise <5% of the national land area (Cumming 1991). Aside from Kruger, no other suitable reserves of sufficient size exist in South Africa to hold a second viable population of wild dogs. The current conservation focus is the creation of a meta-population, based on the reintroduction of packs into a number of geographically isolated reserves, with the aim of establishing a minimum of nine packs within 10 years (Mills et al. 1998). To date, wild dogs have been reintroduced into four state-owned protected areas: Hluhluwe-Umfolozi Park; Madikwe Game Reserve; Marakele National Park; Pilanesberg National Park, and two privately owned protected areas - Karongwe Game Reserve, and Venetia-Limpopo Nature Reserve. In future, expansion of the meta-population is likely to depend increasingly on privately owned reserves.

The 'meta-population management plan' (Mills et al. 1998) is a management intensive process requiring significant logistical investment, particularly during the initial establishment phase of each sub-population (Chapter 5). Six years into the plan, there is a need to assess the efficacy and cost efficiency of this strategy, and to predict the cost efficiency of expanding the meta-population to incorporate private nature reserves.

In addition to the Kruger population and the meta-population, a third population of wild dogs occurs on ranchland, outside state and private nature reserves in South Africa, comprising ~ 76 individuals in ~ 17 packs and dispersing groups (Chapter 2). Focal areas of wild dog activity outside protected areas in South Africa include the ranching areas adjacent to the western border of Kruger, and the ranching areas along the Limpopo River (Chapter 2). Changing land use patterns and an increase in game ranching in South Africa has resulted in increasing potential for conserving wild dogs on ranchland. Given the shortage of large protected areas of suitable habitat type, ranchland is potentially important for the expansion of the South African wild dog population. This potential merits an assessment of the predicted cost efficiency of conserving wild dogs in this land tenure category.

The objectives of this study were twofold. 1) To determine the amount of money spent on wild dog conservation in South Africa over a period of five years (1997 - 2001, the first five years of the meta-population management plan), to determine the source of funding, and to document how it was spent. 2) To assess the cost efficiency of current conservation efforts involving wild dogs - within a large protected area (Kruger), and through the establishment of the meta-population, relative to two potential future conservation options. a) The expansion of the meta-population onto privately owned nature reserves. Here, it was assumed that wild dogs are absent prior to reintroduction, and are prevented from leaving the reserve post release by the presence of perimeter fencing. b) Conserving naturally occurring wild dogs *in situ* on privately owned livestock / game ranchland. Although some of the ranches in such an area may in fact be private nature reserves, this scenario was distinguished from the reintroduction scenario by the fact that wild dogs occur naturally, without requiring reintroduction, and that wild dogs are able to pass between ranches, due to the absence of predator proof fencing between properties. The purpose of this assessment was to provide guidelines for the use of donor funding, and to focus conservation efforts.

6.2 Methods

6.2.1 Expenditure on wild dog conservation in South Africa (1997 – 2001)

Wild dog stakeholders in South Africa (Table 6.1) were asked to how much money was spent on activities related to wild dog conservation, when it was spent, on what it was spent and from whom was it received between 1997 and 2001. 'Stakeholders' included agencies involved in the conservation of wild dogs within Kruger and the meta-population reserves, provincial nature conservation representatives responsible for predators occurring outside of state and provincial parks, and researchers. Expenditures by captive breeders associated with the provision of wild dogs for reintroduction programmes were also documented. Budget records were obtained where possible, and alternatively the costs of activities conducted during the five-year period were estimated. The results presented should be considered minimum expenditure estimates.

Table 6.1 Stakeholders contacted for the collation of records of expenditure made on wild dog conservation during 1997 - 2001

Stakeholder	Involvement
De Wildt Cheetah Breeding Centre	Provision / transport of wild dogs for reintroduction
Hluhluwe-Umfolozi Park	Reintroduction site
Karongwe Game Reserve	Reintroduction site
Kwa-Zulu Natal Wildlife	Reintroduction at HUP [a]
Limpopo Nature Conservation	Capture and transport of wild dogs on ranchland
Madikwe Game Reserve	Reintroduction site
North West Parks Board	Reintroductions at Madikwe and Pilanesberg
Pilanesberg National Park	Reintroduction site
South African National Parks	Kruger population, professional assistance
University of Pretoria	Research
Venetia Limpopo Nature Reserve	Reintroduction site

[a] Hluhluwe-Umfolozi Park.

Expenditure records were converted into 2002 US$ figures, based on the Consumer Price Indices published by the South African Reserve Bank, and the mean US$ / South African Rand exchange rate for the first sixth months of 2002 ($1 = ZAR 10.99).

6.2.2 Cost efficiency indices

The following equation was derived to calculate the cost efficiency of the conservation of wild dogs under various scenarios:

$$CEI = \frac{100,000}{7}\left[\frac{Packs_1}{\left(\frac{C_1}{(1+r)}\right)} + \frac{Packs_2}{\left(\frac{C_2}{(1+r)^2}\right)} + \frac{Packs_3}{\left(\frac{C_3}{(1+r)^3}\right)} + \frac{Packs_4}{\left(\frac{C_4}{(1+r)^4}\right)} + \frac{Packs_5}{\left(\frac{C_5}{(1+r)^5}\right)} + \frac{Packs_5}{\left(\frac{\frac{C_5}{r}}{(1+r)^5}\right)} + \frac{Packs_5}{SSC}\right]$$

CEI represents the cost efficiency index, conceptually based on wild dogs conserved / $100,000 spent, adjusted for time through discounting. $Packs_1 - Packs_5$ represent the number of packs in the population resulting from a given strategy in years 1 - 5, while $Packs_5$ also represents the predicted population size in perpetuity, assuming that the number of packs will remain, or be managed to stay at this size. $C_1 - C_5$ are the costs over five years of a conservation option, while C_5 also represents continuing costs in perpetuity, assuming that the annual costs in perpetuity will be equal to the costs in year five. Costs of a conservation programme are likely to vary for the first few years, and it was felt that after five years, the costs of a conservation strategy would stabilise. *SSC* is the sum of the start up costs associated with a conservation programme, and *r* is the discount rate, based on the average long-term South African Government Bond rates for the first six months of 2002.

This formula calculates a CEI for each of five years using packs and costs in each year, a CEI for the costs of maintaining a stable wild dog population with a known number of packs in perpetuity, and an index of the perpetual number of packs to the initial costs. Each individual CEI is then averaged into the overall CEI, which is multiplied by 100,000 to yield packs / $100,000.

The cost efficiency of conserving wild dogs within a large protected area (Kruger) was calculated slightly differently. Wild dogs have been present in Kruger since the inception of the park, and therefore 'start up costs' were not incurred, and thus excluded from the equation. In addition, significant costs are incurred every five years in Kruger as a result of the five yearly wild dog photographic census, and the equation was modified to account for this, assuming for tractability that one-fifth of the five-yearly cost occurs each year.

$$CEI = \frac{100,000}{6}\left[\frac{Packs_1}{\left(\frac{C_1}{(1+r)}\right)} + \frac{Packs_2}{\left(\frac{C_2}{(1+r)^2}\right)} + \frac{Packs_3}{\left(\frac{C_3}{(1+r)^3}\right)} + \frac{Packs_4}{\left(\frac{C_4}{(1+r)^4}\right)} + \frac{Packs_5}{\left(\frac{C_5}{(1+r)^5}\right)} + \frac{Packs_5}{\left(\frac{\frac{C_5 + \frac{\text{Five-year cost}}{5}}{r}}{(1+r)^5}\right)}\right]$$

6.2.3 Cost efficiency of conserving wild dogs within a large protected area (Kruger)

The cost efficiency of conserving wild dogs within a viable population was estimated, using the expenditure made on wild dogs in Kruger during 1997 – 2001, and the average number of packs counted in the last three photographic censuses in Kruger (28 packs, Maddock & Mills 1994; Wilkinson 1995; Davies 2000). The average number of packs was assumed to be a more accurate representation of the Kruger population than the latest estimate (21 packs), which is believed to constitute an unusually low population size resulting from poor hunting success, probably related to high rainfall (Davies 2000).

6.2.4 Cost efficiency of the establishment of the meta-population

Costs associated with the establishment and maintenance of sub-populations within the meta-population typically include: upgrading of perimeter fencing; upgrading of holding facilities; capture and transport of founders; veterinary costs; feeding the dogs in holding facilities; purchasing monitoring equipment; and monitoring. Wild dogs were reintroduced at Hluhluwe-Umfolozi Park and Madikwe Game Reserve in 1981 and 1994 respectively, and changing personnel prevented the collection of data on the initial reintroduction costs. Consequently, the costs of the initial reintroduction at these reserves were assumed to equal the average initial costs associated with reintroductions undertaken between 1997 and 2001 (Karongwe Game Reserve, Pilanesberg National Park and Venetia-Limpopo Nature Reserve). The costs of predation by wild dogs within the meta-population were not estimated because to date, no donor funding has been provided to compensate host reserves for these costs.

Records from the minutes of Wild Dog Advisory Group-South Africa (WAG-SA) meetings were used to document annual wild dog population sizes within each of the reserves in the meta-population. The total number of packs within the meta-population increased from 3 in 1997 and 1998, to 5 in 1999, 6 in 2000, 8 in 2001, and 10 in 2002. It was assumed that the 2002 population size (10 packs) within the meta-population represents the stable population size for the five reserves into which wild dogs had been reintroduced by that year.

6.2.5 Cost efficiency of the expansion of the meta-population through reintroduction into private nature reserves

In private nature reserves, predation is likely to result in real costs, as prey killed by wild dogs could otherwise be used for hunting or live capture and sale. In light of this, CEIs for the reintroduction of wild dogs onto private land were estimated, incorporating the costs associated with predation. It was assumed that the establishment costs and annual maintenance costs would equal the mean costs recorded at existing meta-population reserves.

Given a high (but within observed limits) annual probability of survivorship of: 0.8 for adult wild dogs; 0.7 for sub-adults; 0.7 for pups and a mean litter size of nine, it is estimated that an average Kruger pack size of five adults and two sub adults reintroduced into a meta-population reserve could potentially increase in size to 20 individuals within five years (Fuller at al 1992). This rate of increase has not been observed in the meta-population however, and for the purposes of this study, it was assumed that a reintroduced pack of seven individuals would increase to the mean 2002 population size observed across reintroduced sub-populations (~ 13 adult and sub adult dogs) in two packs within the first year, and then remain at this level. Although the number of dogs is likely to fluctuate, 13 dogs was used as an average figure for the purposes of calculating costs. Three cost scenarios were presented to allow for variation in the extent to which predation by wild dogs would result in financial loss: a) where the value of all animals killed by wild dogs is fully compensated for; b) where half of prey killed is compensated for (given reduced intensity hunting), and; c) where predation results in no cost. In addition, cost estimates were made for two different prey-profiles, as observed in two parts of South Africa in which the reintroduction of wild dogs is likely to occur, northeastern South Africa and eastern South Africa, as recorded by Mills & Gorman (1997) and Kruger et al (1999) respectively. The costs of predation were estimated as in Chapter 5.

6.2.6 Cost efficiency of the conservation of wild dogs occurring on ranchland

If donor funding was utilised for the conservation of wild dogs on ranchland, it is assumed that the dogs would be monitored in order to help prevent persecution, and to assist with the allocation of funds for the compensation of land owners for losses due predation by wild dogs. Although ranchers attitudes towards wild dogs are variable, most negative attitudes are based on the perceived or real costs associated with their presence (Chapter 4). It is therefore assumed that the provision of compensation for losses due to predation by wild dogs would be sufficient incentive to encourage landowners to tolerate the species on their land. The costs of capturing wild dogs to attach telemetry equipment were estimated as in Chapter 5. It was assumed that initially, three of the dogs would be radio-collared. Subsequent costs include those associated with intensive monitoring, re-capture to add collars and the cost of compensating for predation by wild dogs. It was assumed that three dogs would be immobilised annually to replace radio-collars and add collars to young individuals. With adequate habituation following release, wild dogs can be re-captured from the reintroduction site by darting from a vehicle and the costs will include vehicle usage, veterinary labour, and capture drugs. Kilometers driven was used as a index of monitoring effort - it was assumed that monitoring is conducted at a rate equal to that at Venetia-Limpopo Nature Reserve in the first year post-release (4,000 km monthly). The costs of re-capturing wild dogs following habituation, telemetry equipment, vehicle use and salaries associated with monitoring were estimated in the same way, and derived from the same sources as in Chapter 5.

Natural habitat is highly fragmented in South Africa (Chapter 2) and there is a limit to the number of wild dogs that could potentially be conserved in a given area on ranchland. The average number of resident packs occurring in each of the two areas in which wild dogs are most regularly sighted on private land in South Africa during 1996 and 2002 was 2.5 packs, and cost estimates were made for a stable sub-population size of three packs of the average size observed on ranchland (~ 7 dogs / pack, Chapter 2). Recently, wild dogs naturally re-colonised a game ranching area in Zimbabwe (where ranches have been consolidated into a collaborative nature reserve) and exhibited high rates of population increase (Pole 1999). It was assumed that an average newly formed pack (6 dogs) colonising an area of ranchland with adequate prey availability would exhibit a rapid increase in numbers, from 6 in 1 pack, to 11 in two packs, 15 in two packs, 18 in three packs, to 21 in three packs in years 1 to 5, respectively, given published survivorship rates (Fuller et al. 1992). For the sake of cost, and cost efficiency calculations, it was assumed that the population size would remain at 21 individuals in 3 packs. The same predation cost scenarios were presented as for the private nature reserve reintroductions, with one difference. Wild dogs on private land are likely to come into contact with livestock and each prey-profile is assumed to include the same proportion of cattle (32.2%) observed in the sole published study of wild dogs in a ranching area (Rasmussen 1999).

6.3 Results

6.3.1 Expenditure on wild dog conservation (1997 - 2001)

An estimated \$372,297 was spent on the conservation of wild dogs in South Africa between 1997 and 2001, at an average of \$74,459 per annum. Of this \$270,117 (72.6%) was spent specifically on the meta-population, \$57,863 (15.5%) on wild dogs in Kruger and \$33,942 (9.1%) on wild dogs on ranchland (Table 6.2). The remainder was spent on wild dog research not specifically related to any of the three populations (Frantzen et al. 2001; Knobel & du Toit 2003).

NGOs provided the most funding for wild dog conservation in South Africa during 1997 - 2001 (39.9%), followed by South African state agencies (36.8%), private donors

Table 6.2 Expenditure on the conservation of the three sub units of the South African wild dog population during 1997 – 2001, in 2002 US$ (ZAR in parentheses)

Sub unit	1997	1998	1999	2000	2001	Total
Kruger	7,505 (82,480)	7,305 (80,282)	24,820 (272,772)	9,829 (108,021)	8,404 (92,360)	57,863 (635,915)
Meta-population						
Hluhluwe	13,484 (148,189)	246 (2,704)	14,204 (156,102)	15,644 (171,928)	13,499 (148,354)	57,077 (627,276)
Karongwe	-	-	-	-	29,295 (321,952)	29,295 (321,952)
Madikwe	9,719 (106,812)	4,321 (47,488)	4,408 (48,444)	7,735 (85,008)	14,375 (157,981)	40,558 (445,733)
Pilanesberg	-	-	39,783 (437,215)	3,278 (36,025)	4,860 (53,411)	47,921 (526,651)
Venetia	-	-	-	-	79,750 (876,453)	79,750 (876,453)
[a] Miscellaneous	4,835 (53,137)	761 (8,364)	723 (7,946)	686 (7,539)	8,511 (93,536)	15,516 (170,522)
On ranchland	4,155 (45,663)	2,131 (23,420)	5,852 (64,312)	5,434 (59,720)	16,370 (179,906)	33,942 (373,021)
TOTAL	39,698 (436,281)	14,764 (162,258)	89,790 (986,791)	42,606 (468,241)	175,064 (1,923,953)	361,922 (3,977,524)

[a] Including the costs of a workshop at which the meta-population management plan was conceived (Mills et al. 1998), the costs of Wild dog Advisory Group-SA meetings, and the costs of purchasing of founder dogs for reintroductions.

(20.8%) and universities (2.5%). The majority of expenditure on the Kruger population (Figure 6.1) was provided by state agencies (65.8%), the remainder being provided by NGOs (34.2%). NGOs provided most of the money spent on the meta-population (44.5%), followed by private donors (27.9%), state agencies (26.8%), and universities (0.8%). The majority of the money spent on wild dogs on ranchland was provided by state agencies (71.9%), followed by NGOs (24.4%), and universities (3.7%).

In terms of money spent on the Kruger population, 64.0% of the money was spent on research, 34.1% on a photographic census of the population, 1.0% on attending meetings, and the remaining 0.9% on the capture and veterinary care of wild dogs, primarily for the removal of snares. For the meta-population (Figure 6.2), most was spent on monitoring and research (48.2%), feeding dogs in holding facilities (13.1%) and the upgrading of perimeter fencing (13.0%, more details in Appendix H). Of the money spent on wild dogs on ranchland, 39.7% was spent removing "problem animals", 29.0% was spent on research, 23.3% was spent on attending ranchers complaints, and 8.0% was spent by provincial nature conservation representatives attending wild dog-related meetings.

6.3.2 Cost efficiency of conserving wild dogs within a large protected area (Kruger)

The mean annual costs associated with conserving wild dogs in a large protected area are \$11,573 (Table 6.3). Assuming that the mean population size of 28 packs of dogs within Kruger between 1988 – 2000 represents the stable long term population size, the cost efficiency of conserving wild dogs within Kruger is estimated to be 449 packs / \$100,000 (Table 6.4). Conserving wild dogs within a large protected area is estimated to be more

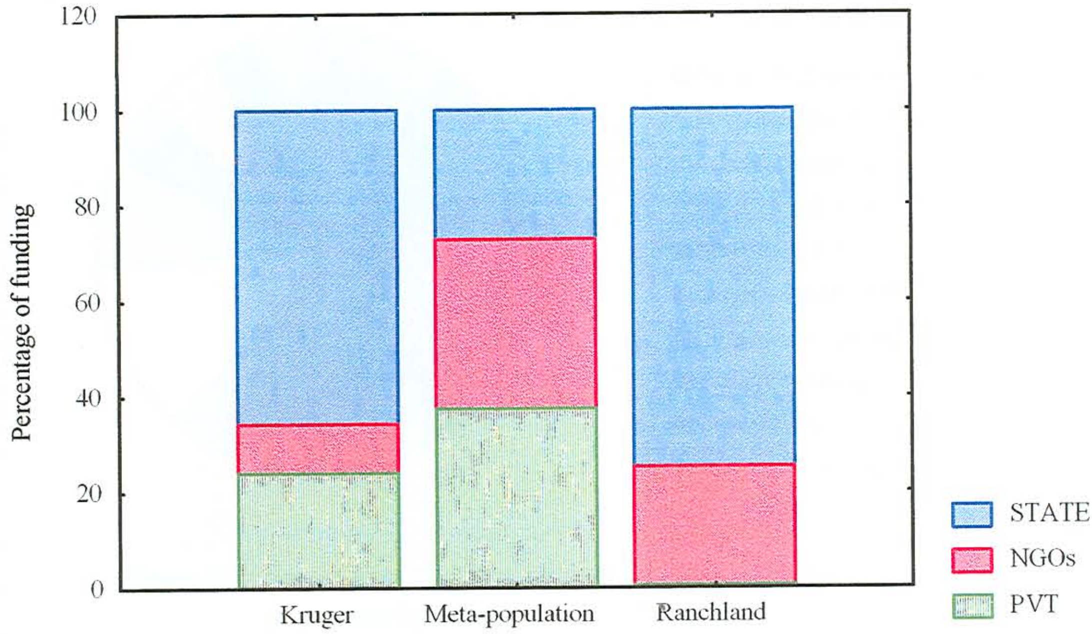

Figure 6.1 Source of expenditure (STATE - state agencies, NGOs and PVT - private companies) for each sub unit of the South African wild dog population during 1997 - 2001

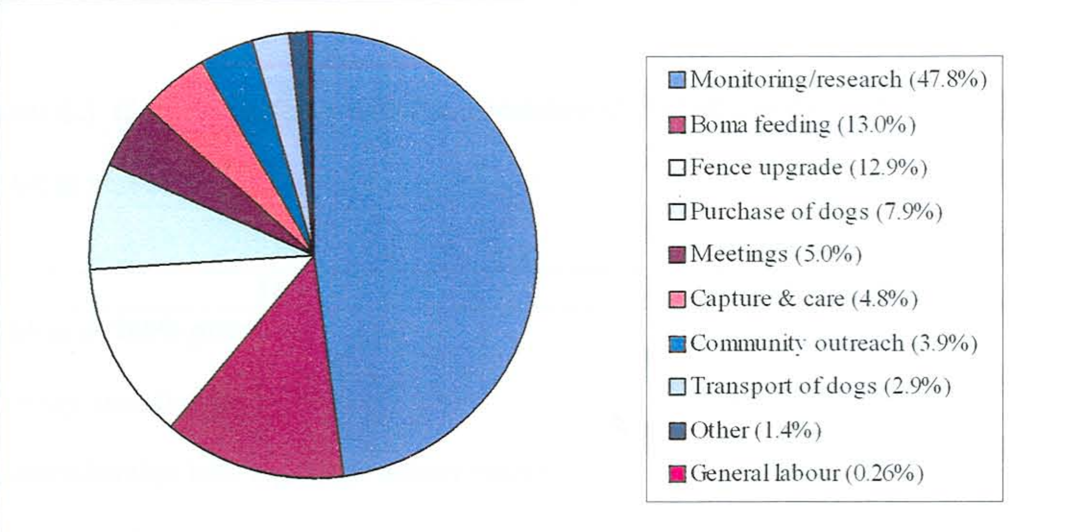

Figure 6.2 Breakdown of expenditure on the wild dog meta-population by activity (1997 - 2001)

Table 6.3 Cost estimates used for the calculation of cost efficiency indices, in 2002 US$ (ZAR in parentheses)

Item	Costs	
Within a viable population		
Average annual costs [a]	11,573	(127,187)
Reintroduction into a private nature reserve		
Initial costs [b]	36,880	(405,311)
Annual running costs [c]	10,554	(115,988)
Predation [d]		
ESA prey-profile [e], all prey compensated	101,762	(1,118,364)
ESA prey-profile, half prey compensated	50,881	(559,182)
NESA prey-profile [f], all prey compensated	17,761	(195,193)
NESA prey-profile, half prey compensated	9,563	(105,097)
Conservation of wild dogs on ranchland		
Initial costs		
First (helicopter-assisted) capture	1,980	(21,760)
Purchase of telemetry equipment	1,592	(17,496)
Average annual running costs		
Capture (darting from a vehicle)	1,012	(11,122)
Purchase of additional radio collars	721	(7,924)
Employee salary	4,648	(51,082)
Vehicle devaluation and maintenance	9,001	(98,921)
Predation costs		
ESA prey-profile [e], all prey compensated	115,761	(1,272,213)
ESA prey-profile, half prey compensated	57,881	(636,112)
NESA prey-profile [f], all prey compensated	22,883	(251,484)
NESA prey-profile, half prey compensated	11,942	(131,243)

[a] Equal to the average annual expenditure upon wild dogs in Kruger during 1997 – 2001.

[b, c] Equal to the average costs associated with the initial reintroduction and annual maintenance of wild dogs in the meta-population to date.

[d] Assuming the number of dogs is equal to that in year five.

[e] Eastern South Africa.

[f] Northeastern South Africa.

Table 6.4 Cost efficiency indices (dogs / $100,000) of conserving wild dogs under three conservation programmes in perpetuity

Scenario	CEI
Within a viable population	449
Establishment of the meta-population so far	23
Reintroduction into private reserves	
ESA prey-profile [a]	
All prey compensated	3
Half prey compensated	4
NESA prey-profile [b]	
All prey compensated	8
Half prey compensated	11
Zero predation costs	19
***In situ* on ranchland**	
ESA prey-profile [a]	
All prey compensated	14
Half prey compensated	16
NESA prey-profile [b]	
All prey compensated	19
Half prey compensated	22
Zero predation costs	27

[a] Eastern South Africa.
[b] Northeastern South Africa.

cost efficient than the meta population management plan by 20 times, than the reintroduction of dogs into a private nature reserve by 24 – 150 times, and than conserving wild dogs on ranchland by 17 – 32 times (Table 6.4).

6.3.3 Cost efficiency of the establishment of the meta-population

The meta-population increased from 19 individuals in three packs in 1997, to 54 sub adults and adults in 10 packs prior to the denning season in 2002, and the population target for the meta-population has been achieved in just over half of the time set aside for this purpose (Mills et al. 1998). The changes in dog and pack numbers by reserve during 1997 – 2001 were as follows: Hluhluwe-Umfolozi Park (13 adults and sub adults in two packs, down to seven sub adults and adults in two packs); Karongwe Game Reserve (zero up to four adults and sub adults in one pack); Madikwe Game Reserve (six adults and sub adults in two packs, up to 17 in three packs); Pilanesberg National Park (zero up to 13 adults and sub adults in two packs); Venetia Limpopo Nature reserve (9 adults and sub adults were released from holding facilities in January 2002, now two packs). An additional 16 wild dogs were released into Marakele National Park in May 2003. Assuming the number of wild dogs within the meta-population (excluding Marakele) stabilises at the 2002 population size of 54 sub adults and adults in 10 packs, this represents a cost efficiency of 23 packs / $100,000 (Table 6.4). The meta-population management plan so far is predicted to be more cost efficient than expanding the meta-population onto private nature reserves, within the predicted range of CEIs for the conservation of dogs on ranchland, and substantially less cost efficient than the conservation of dogs within a large protected area.

6.3.4 Cost efficiency of the expansion of the meta-population through reintroduction into private nature reserves

Mean expenditure on initial reintroductions of wild dogs into meta-population reserves over the last five years was $36,880 (range $5,372 - $79,750; n = 3 reintroductions), while the mean annual maintenance expenditure was $10,554 (range $4,948 - $56,862; n = 5 reintroductions, Table 6.2). Predicted annual costs of predation vary depending on the observed prey-profile and the proportion of prey killed that is compensated for (Table 6.3). The estimated CEI of reintroducing and conserving wild dogs within a private nature reserve is: 19 packs / $100,000 where predation results in no costs; 11 packs / $100,000 given a northeastern South African prey-profile, where half of prey killed is compensated for; 8 packs / $100,000 given a northeastern prey-profile where all prey is compensated for; 4 packs / $100,000 given an eastern South African prey-profile where half of prey killed is compensated for; and 3 packs / $100,000 given an eastern South African prey-profile where all prey are compensated for (Table 6.4). The expansion of the meta-population through reintroduction onto private nature reserves is predicted to be the least cost efficient strategy of those considered.

6.3.5 Cost efficiency of the conservation of wild dogs on ranchland

The costs of establishing a conservation initiative involving wild dogs on ranchland are <10% (9.7%) of those of reintroducing a pack of wild dogs into a reserve (Table 6.3). Average annual costs associated with predation by a sub-population of wild dogs are estimated to be 79.3% greater under an eastern South African prey-profile than under a northeastern South African prey-profile. The estimated CEI of reintroducing and conserving wild dogs on ranchland is: 27 packs / $100,000 where predation results in no costs; 22 packs / $100,000 given a northeastern South African prey-profile where half of prey killed is compensated for; 19 packs / $100,000 given a northeastern prey-profile where all prey is compensated for; 16 packs / $100,000 given an eastern South African prey-profile where half of prey killed prey is compensated for; and 14 packs / $100,000 given an eastern South African prey-profile where all prey are compensated for (Table 6.4)

6.4 Discussion

6.4.1 Expenditure on wild dog conservation (1997 - 2001)

After many years of being overshadowed by Africa's better-known carnivores, wild dogs have received an increasing amount of attention from researchers and donors in recent years (Creel & Creel 2002). This interest was reflected in the amount spent on their conservation in South Africa between 1997 and 2001. Over $370,000 was spent, with donors including a variety of NGOs, private companies and state agencies. However, although substantial, this amount is dwarfed by expenditure estimates for other high profile carnivore species. An estimated $6 million is spent annually on tiger *Panthera tigris* conservation, and over $2 million spent annually within the Indian subcontinent alone (Christie 2001). In the USA, $350,000 was spent on wolf conservation in a single state (Minnesota) in 1998 (Mech 1998). Nonetheless, funding received for wild dog conservation in South Africa is increasing, and for this support to continue, the use of funds must be shown to be effective (Christie 2001).

Six years after the initiation of the meta-population management plan (Mills et al. 1998) the target population of nine packs was exceeded, and wild dogs have been successfully established and maintained in five reserves, with an additional reintroduction having been conducted in May 2003 at Marakele National Park. In addition, the Kruger wild dog population has been closely monitored. During 1988 and 2000, the Kruger population has fluctuated widely, increasing by 17.8% between 1988 - 1995, and then declining by 59.2% between 1995 - 2000 (Maddock & Mills 1994, Wilkinson 1995, Davies 2000). These population fluctuations stress the need for continued monitoring, and continued investment in the meta-population as an 'insurance policy'. It is reasonable to say that wild dog conservation in South Africa has been effective within the limits of set targets (Mills et al. 1998). Beyond these limits, however, little has been done to improve the conservation status of wild dogs, and very little funding was directed at the population occurring on ranchland. Money that was spent was directed primarily at the removal of 'problem' packs and consequently had a negative effect on the conservation of wild dogs on ranchland.

6.4.2 The cost efficiency of wild dog conservation

Maintaining large protected areas represents the single most important strategy for wild dog conservation (Woodroffe & Ginsberg 1997a), and is the most cost efficient way in which wild dogs can be conserved in South Africa. Most of the costs associated with conserving wild dogs under this scenario are indistinguishable from the costs of maintaining a large protected area in general, and very little specific expenditure is required. Furthermore, much of the expenditure on wild dogs within Kruger (for example the photographic census) is not vital for the persistence of the population, and the cost efficiency of this strategy is potentially much greater. The expansion of several South African protected areas across national boundaries to create large transfrontier parks is proposed, and this creates potential for the expansion of the Kruger and Gona-re-zhou National Park (Zimbabwe) wild dog populations into Mozambique (Great Limpopo Transfrontier Park), and the establishment of viable populations in the proposed Limpopo / Shashi and Lubombo transfrontier conservation areas (www.peaceparks.org). Although the initial reintroduction of wild dogs into a large protected area would likely require more funds than necessary for the reintroduction of dogs into a meta-population reserve due to the necessity for a larger founder population for the creation of a population viable in the absence of artificial immigration, the ongoing costs would likely be negligible. Given the potential for establishing additional viable populations at a relatively low cost, it is suggested that donor funds be used to reintroduce wild dogs into the proposed Limpopo / Shashi and Lubombo transfrontier conservation areas as soon as they are established.

The meta-population management plan has been substantially less cost efficient than the conservation of wild dogs in large protected areas, due to the logistical difficulty associated with the reintroduction process. The expansion of the meta-population to include additional private nature reserves is likely to be less cost efficient still. So far, the reserves into which wild dogs have been reintroduced (largely state-owned) have absorbed the costs of predation by wild dogs post-release. The costs of predation by wild dogs are potentially very high (Chapter 5) and private nature reserves may not be willing to accept these losses in the absence of compensation. Such compensation, in addition to the high start-up and maintenance costs would reduce the cost efficiency of wild dog reintroductions below that of competing conservation strategies. Furthermore, most of the reserves into which dogs have been reintroduced to date have had to invest relatively little in upgrading perimeter fencing or holding facilities due to existing high quality infrastructure. Upgrading standard game fencing to the specifications required for wild dogs is extremely costly (Chapter 5), and if the meta-population is expanded onto reserves without pre-existing predator proof fencing, the cost efficiency would decline further.

Under certain conditions, however, private nature reserve owners may be encouraged to reintroduce wild dogs at their own cost. Ecotourism is the most profitable land use on reserves of a size sufficient for wild dog reintroductions (Falkena 2000) and under these conditions the financial impact of predation by wild dogs is likely to be negligible. In addition, high quality fencing and boma facilities are likely to be already present, due to the importance of other carnivore species such as lions for attracting visitors to a reserve (Vorhies & Vorhies 1993). Furthermore, it is likely that as the methodology associated with reintroducing wild dogs and maintaining them post release improves, the process will become more efficient and costs will decline. Some of the costs incurred during reintroductions to date (such as holding dogs in captivity for lengthy periods, and extensive community outreach programmes) are not vital for the success of a reintroduction programme and could be excluded. Finally, the potential financial benefits associated with wild dog-based ecotourism are substantial ($11,000 - $64,000 / pack /

year), and sufficient to exceed the costs associated with reintroduction programmes under certain conditions (Chapter 5). In keeping with this, the agency responsible for the management of the meta-population (Wild Dog Advisory Group-SA) has received several applications for wild dog reintroductions from private nature reserve owners. Consequently, it is suggested that the expansion of the meta-population be limited to private nature reserves willing to carry the costs. Although some donor funding would still be required, for the provision of suitable founder animals and to provide technical assistance pre, and post release, donor funding requirements would be greatly reduced.

The conservation of wild dogs on ranchland has received very little attention, and yet is predicted to be of similar or greater cost efficiency than the current meta-population management plan under realistic scenarios, and substantially more cost efficient (by up to 5 times) than the expansion of the meta-population onto private nature reserves. The cost efficiency of conserving wild dogs on ranchland is likely to be closer to (or higher than) the higher estimates made in this chapter (22 – 27 packs / $100,000) for several reasons. First, impala *Aepyceros melampus* and kudu *Tragelaphus strepsiceros* are the most important components of the northeastern prey profile (Mills & Gorman 1997), and are the most common ungulates in most areas in which wild dogs occur on ranchland. Second, in some parts of South Africa, up to 33% of ranches are involved in ecotourism-based land uses (Chapter 4), and under these conditions, the costs of predation by wild dogs are likely to be much reduced or absent. Finally, there is scope for offsetting some or all of the costs of conserving wild dogs on ranchland with ecotourism-benefits, reducing dependency upon donor funding, and increasing cost efficiency further (Chapter

5). Indeed, a more sustainable and cost efficient strategy than compensating landowners for losses caused by predation would be to assist ranching communities to establish ecotourism schemes involving wild dogs, to enable wild dogs to effectively 'pay for their own conservation'. There are difficulties associated with using compensation as a conservation management tool on ranchland. Donor funding would be required indefinitely, and potentially to an increasing extent as populations of wild dogs on ranchland spread. The cessation of compensation at any point may result in a reversal of conservation achievements. Compensating ranchers for prey killed by wild dogs likely represents a worst case cost efficiency scenario for conserving wild dogs *in situ* on ranchland. Removing the costs of compensation, and replacing them with the costs of educational programmes and technical assistance for the establishment of wild dog-ecotourism operations would increase the cost efficiency of this conservation option markedly.

The best prospects for conserving wild dogs on ranchland occur where neighbouring ranchers have cooperated to remove fences and create large collaborative nature reserves. Under these circumstances, ranchers are typically more positive towards wild dogs, ecotourism based land uses are prevalent, predation by wild dogs is likely to result in little or no cost, and the scope for wild dog-based ecotourism is greatest (Chapter 4).

There is a ready supply of founder wild dogs on ranchland (Chapter 2), and large areas of suitable habitat as a result of the increasing prevalence of game ranching (Lambrechts 1996; van der Waal & Dekker 2000). Given adequate prey availability and sufficient protection, wild dogs have the potential to increase in numbers rapidly following the re-colonisation of private land (Pole 1999) and there is reason to be optimistic that realistic conservation targets on ranchland in South Africa could be reached. Conserving wild dogs on ranchland has a potentially important role in increasing numbers and geographic distribution, and in providing buffers for populations occurring in adjacent protected areas (Woodroffe & Ginsberg 1998).

The cost efficiency approach adopted in the present study has wide application for other threatened species. There is increasing competition for funds between species, and between conservation projects within species, and funding agencies place increasing emphasis on 'value for money' in project choice (Restani & Marzluff 2001). Efforts to secure donor funding for endangered species are frequently hampered by a lack of good cost information (Wilcove & Chen 1998). Cost efficient conservation programme design is likely to improve the chances of financial support (Moran et al. 1997), maximise results of conservation efforts, and can benefit other species by increasing the availability of funds. This is well illustrated by the situation in North America, where improved cost efficiency in conservation programmes for a few species has the potential to benefit many. In 1995, of the $348 million spent on endangered species in the USA, >50% went to 10 species (Baker 1999). In sum, cost efficient conservation is vital to minimise the discrepancy between current global spending and the funding requirements of all threatened species (Balmford et al. 2003).

Wild dogs were an ideal species with which to investigate the role of donor funding in the conservation of an endangered species in Africa. They have been the focus of a large number of studies scattered across Africa (Creel & Creel 2002), and focused conservation efforts in South Africa. Donor support for wild dog conservation in South Africa has been reflected in the rapid attainment of conservation targets. Donor funding is most effective for species threatened by human activities (Miller et al. 2002). Human-related mortality is the primary cause of decline in wild dog numbers across Africa (Woodroffe & Ginsberg 1997b), and consequently, donor funding has the potential to improve the conservation status of species throughout it's range.

In South Africa, it is suggested that monitoring efforts be continued in Kruger, and that donor funding be used to establish wild dog populations in proposed transfrontier parks as soon as they are established. In addition, it is suggested that donor funding be directed towards the conservation of wild dogs on ranchland, and the maintenance of the meta-population. Expansion of the meta-population should be limited to reserves willing to carry the costs. Consideration of the cost efficiency of conservation options has an important role in guiding future conservation strategies involving wild dogs, and a wide variety of other threatened species in Africa, and elsewhere.

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CHAPTER 7

Summary and conclusions

Expansion of the geographic range of wild dogs *Lycaon pictus* in South Africa is of key importance to reduce the risks of extinction associated with small populations. A proliferation of private nature reserves and game ranches in South Africa (van der Waal & Dekker 2000) has resulted in increasing scope for conserving wild dogs on private land. This chapter summarises a study which: a) assessed the potential for conserving wild dogs outside state protected areas on private land; b) identified conditions under which conservation efforts are most likely to succeed; and c) suggested appropriate conservation strategies.

7.1 Answers to questions addressed in this study

7.1.1 What is the present distribution and population status of wild dogs outside state protected areas in South Africa?

Wild dogs outside protected areas form a more significant component of the South African wild dog population than previously recognised (c.f. Fanshawe et al. 1997). The population of wild dogs occurring outside state protected areas in South Africa is estimated to have fluctuated between 42 and 106 animals during 1996 - 2002. The number of wild dogs resident outside protected areas during this period, ranged between 25 and 67 animals during 1996 - 2001, with an extent of occurrence of 43,310 km^2 and an area of occupancy of 17,907 km^2. Sightings were most commonly reported from the western border of Kruger National Park (henceforth referred to as "Kruger"), along the Limpopo River, and from northern Kwa-Zulu Natal. The majority of sightings of resident wild dogs occurred on game ranches (77.2%), with unmodified land cover (91.2%), at low human densities (19.5 ± 4.77 people / km^2; mean ± S.E.) and close to source populations (86.6 km ± 5.02). Dispersing wild dogs were sighted further from source populations (187 km ± 16.1), in areas of higher human density (47.1 people / km^2 ± 2.24), in more modified habitat (19.6% of sightings occurred in areas of degraded or unsuitable land cover), and in more varied land use (only 32.9% on game ranches).

An estimated 264,900 km^2 of potentially suitable habitat (unmodified natural land cover and <5 people / km^2) for the conservation of wild dogs exists north of the southerly borders of the Free State and Kwa-Zulu Natal provinces. However, habitat fragmentation, the presence of high-speed roads and negative human attitudes are likely to substantially reduce the area available for range expansion. The best scope for range expansion probably exists in Limpopo (46,550 km^2 of potentially suitable habitat), and North West (39,725 km^2 of potentially suitable habitat) due to the prevalence of game ranching (van der Waal & Dekker 2000) and the proximity of source populations. Despite large areas of unmodified habitat with low human densities, the Free State (49,350 km^2 of potentially suitable habitat) and Northern Cape (115,650 km^2 of potentially suitable habitat north of the southern Free State border) provinces probably provide poor prospects for wild dog conservation. In both provinces, wild dogs would likely conflict with sheep farming, and the absence of resident wild dogs in the Kgalagadi Park in the Northern Cape (Fanshawe et al. 1997) suggests that the habitat in the dry west of South Africa is likely sub-optimal for wild dogs.

7.1.2 What are the minimum area and prey requirements for a pack in the areas in which wild dogs occur in South Africa?

An estimated minimum area of 158.5 km^2 is required to support a pack of 12 wild dogs in northern South Africa, compared to 172.8 km^2 in eastern South Africa, and 354.2 km^2 in northeastern South Africa. A pack size of five is the threshold, below which reproductive failure is likely (Courchamp & Macdonald 2001). The area requirements of five wild dogs are estimated to be 65.4 km^2 in northern South Africa, 72.1 km^2 in eastern South Africa and 147.2 km^2 in northeastern South Africa. These results suggest that reserves smaller than those previously utilised might be considered for wild dog reintroductions. These estimates represent the theoretical minimum area necessary to support wild dogs, and observed home range areas under natural conditions are significantly larger (by 1.2 times in eastern South Africa, 2 times in northeastern South Africa, and 3.5 times in northern South Africa). Conservation of wild dogs at the theoretical maximum density would require intensive management of the factors known to limit wild dogs under natural conditions.

7.1.3 What are the attitudes of landowners towards wild dogs, and the reasons for these attitudes in the areas in which wild dogs occur on private land in South Africa?

Wild dogs are the least popular large carnivores among southern African ranchers. Despite this, 52.3% of ranchers indicated that they would like to have wild dogs on their property. Rancher's attitudes varied significantly between sample sites; the highest proportion of ranchers were positive in eastern South Africa (69.2%), followed by in Zimbabwe (66.6%), northeastern South Africa (58.5%) and finally, northern South Africa (24.1%). The proportion of land area occupied by ranchers positive towards wild dogs varies from 75.5% in eastern South Africa, to 68.9% in northeastern South Africa, and 39.4% in northern South Africa.

The most common reasons for negative attitudes towards wild dogs were "they affect my income" (13.5% of ranchers), "they kill a lot / too much game" (13%), "they kill livestock" (12%), and "they chase game and make it wild" (10.6%). The complaints of many ranchers (44.7% of ranchers) were related to economic costs associated with the presence of wild dogs. Conservation initiatives aimed at reducing the costs associated with the presence of wild dogs represent the most direct way in which to improve attitudes. The most common reasons for positive attitudes towards wild dogs were; 'their ecotourism value' (21.6%), 'their ecological role' (13.5%), and 'because wild dogs only pass through the property' (12.5%).

Attitudes are most positive where neighbouring ranchers have cooperated to remove internal fencing to form collaborative nature reserves, and where land use is dominated by ecotourism. Attitudes are likely to be negative where ranches are isolated from neighbours by perimeter game fencing, and where cattle ranching or the consumptive utilisation of wildlife dominate land use.

7.1.4 What are the costs and potential benefits associated with conserving wild dogs within a viable population, through reintroduction into a reserve, and in situ, *on ranchland?*

The costs (estimated present values) of conserving a pack of wild dogs in perpetuity under three scenarios are as follows: 1) $1,850 within a viable population; 2) $78,861 - $856,928 for reintroduction into and maintenance within a private nature reserve utilised for commercial wildlife production (depending primarily upon the need for upgrading perimeter fencing, and the costs of predation by wild dogs post-release); 3) $47,467 - $460,136 on ranchland, depending upon the costs of predation.

The willingness of tourists to pay to go on a guided tour to see wild dogs at a den varied from a minimum of $11 / person / trip, among visitors to a public reserve (Kruger), to $53 / person / trip among visitors to a private nature reserve (Ngala Game Reserve). In the scenario of two trips to a den daily with a safari vehicle capable of carrying nine guests over a period of three months, and a 75% booking rate, potential annual income is $11,110 under the Kruger estimate of willingness to pay, and $63,499 under the Ngala estimate of willingness to pay. The net present value of conserving a pack of wild dogs in a viable population in perpetuity is estimated to be $90,887. Potential annual tourism revenue from a single pack in Kruger ($11,110) is sufficient to offset the mean annual costs of the conservation of the entire Kruger population ($1,712). Given the high volumes of tourists and multiple wild dog packs in Kruger, it is realistic to argue that Kruger could potentially generate sufficient funds from wild dog-based ecotourism to subsidise wild dog conservation efforts elsewhere.

The potential for tourism revenue to offset the costs of the reintroduction and conservation of wild dogs in a private nature reserve is dependent upon predicted tourism benefits, the need for modifications to perimeter fencing, and the post release costs of predation by wild dogs. Tourism revenue is only likely to offset costs if tourist willingness to pay is high, and costs associated with post-release predation by wild dogs are negligible. Under these conditions, tourism revenues have the potential to act as an incentive for private nature reserve owners to reintroduce wild dogs at their own cost. On ranchland, tourism revenue is predicted to offset the costs of conserving wild dogs under most scenarios and has the potential to act as a real incentive for ranchers to conserve wild dogs on their properties.

7.1.5 To what extent has donor funding subsidised wild dog conservation in South Africa in recent years?

An estimated \$372,297 was spent on the conservation of wild dogs in South Africa between 1997 and 2001, at an average of \$74,459 per annum. Of this \$270,117 (72.6%) was spent specifically on the meta-population, \$57,863 (15.5%) on wild dogs in Kruger and \$33,942 (9.1%) on wild dogs on ranchland. NGOs provided the most funding for wild dog conservation in South Africa during 1997 - 2001 (39.9%), followed by South African state agencies (36.8%), private donors (20.8%) and universities (2.5%). The majority of expenditure on the Kruger population was provided by state agencies (65.8%), the remainder being provided by NGOs (34.2%). NGOs provided most of the money spent on the meta-population (44.5%), followed by private donors (27.9%), and state agencies (26.8%). The majority of the money spent on wild dogs on ranchland was provided by state agencies (71.9%), followed by NGOs (24.4%).

The use of donor funding in wild dog conservation over the last five years has been effective within the limits of set targets (Mills et al. 1998). The target size of the meta-population (9 packs) has been exceeded in just over half the proposed time frame (Mills et al. 1998), and the Kruger population has been closely monitored (Davies 2000). Beyond this, however, little has been achieved. Wild dogs occurring on ranchland remain heavily persecuted and remain few in number, with a limited distribution. Adequate protection of these animals is vital to increase the geographic range of wild dogs in South Africa and to protect the integrity of sub-populations occurring within protected areas.

7.1.6 What is the most cost efficient strategy for improving the status of wild dogs in South Africa?

When the cost efficiency of wild dog conservation is considered in terms of packs conserved per $100,000 invested, then conserving wild dogs in large protected areas represents the most cost efficient strategy (449 packs / $100,000). The establishment of the meta-population has been a far less cost efficient conservation strategy, yielding 23 packs / $100,000. If wild dogs are reintroduced into private nature reserves to expand the meta-population, costs may be higher still. The costs of predation by wild dogs are potentially very high, and private nature reserves may not be willing to accept these losses in the absence of compensation. Such compensation, in addition to the high start-up and maintenance costs would reduce the cost efficiency of the reintroduction of wild dogs to an estimated 3 - 11 packs / $100,000 spent, depending upon the predicted costs of predation post-release. The conservation of wild dogs *in situ* on ranchland is predicted to be a more cost efficient strategy (14 - 27 packs / $100,000) due to the absence of the high start up costs associated with wild dog reintroductions.

I suggest that donor funding should be used to reintroduce wild dogs to the proposed transfrontier parks as soon as they are established, to create additional viable populations in large protected areas at relatively low cost. In the meantime, it is suggested that donor funding be used to maintain the current meta-population, and to establish conservation programmes involving wild dogs on ranchland. Expansion of the meta-population should be limited to private nature reserves willing to carry the costs.

7.2 Final conclusions

South Africa has a total free ranging wild dog population of 279 - 307 individuals, occurring in three separate distributions: Kruger (57.7% of total population); outside protected areas (24.7% of total population), and the protected meta-population (17.6% of total population – based upon the 2002 pre-denning season population estimate). Kruger supports South Africa's sole viable population and forms the core of current conservation efforts. The meta-population represents an attempt to create a second viable population, and comprised an additional 54 adults and sub adults, in 10 packs, in five sub populations prior to the denning season in 2002. An additional reintroduction has since occurred during May 2003, at Marakele National Park (16 wild dogs released). Wild dogs outside protected areas form a more significant portion of the population than had been realised. High levels of persecution, however, prevent this population from expanding to fill large areas of potentially suitable habitat. Assuming that the area of suitable habitat for wild dogs equals the area of unmodified land with fewer than five people / km^2 in Kwa-Zulu Natal, Limpopo, and North West (88,750 km^2, conservatively assuming that all habitat in the Northern Cape and Free State provinces is unsuitable), the potential population size of wild dogs occurring outside protected areas is 178 individuals (~ 18 packs) given a density equal to the lowest observed in a protected area (two dogs / 1000 km^2, Fuller et al. 1992), or 1,482 individuals (~ 148 packs) given a density equal to the minimum density observed in Kruger (16.7 dogs / 1000 km^2, Mills & Gorman 1997). The results of my study suggest that conservation efforts should be focused as discussed in the following paragraphs.

7.2.1 Large protected areas

The monitoring of the Kruger wild dog population should be continued, and donor funding should be used to establish wild dog populations in the proposed Limpopo / Shashi and Lubombo transfrontier conservation areas (www.peaceparks.org) as soon as they are established.

7.2.2 The meta-population

The expansion of the meta-population should be limited to state reserves, and private nature reserves willing to carry the costs. Potential revenue from wild dog-based ecotourism is substantial under certain conditions, and it is realistic to expect that the expansion of the meta-population could be achieved in this manner, with minimal donor funding. Private nature reserve owners should be provided with assistance to establish, and market wild dog-based ecotourism enterprises. Reserves smaller than those utilised to date should be considered for wild dog reintroduction.

7.2.3 On ranchland

Donor funding should be directed towards the conservation of wild dogs on ranchland, and be used to assist ranchers to establish ecotourism operations involving wild dogs on their land, so as to offset the costs associated with their presence. Collaborative nature reserves, and areas dominated by ecotourism-based land uses represent ideal targets for conservation initiatives. In areas where wild dog-based ecotourism is not possible, or unlikely to exceed the costs, compensation schemes might be considered as a means of reimbursing landowners for losses caused by wild dogs. Prior to the onset of such a scheme, however, careful consideration should be given to potential difficulties and problems. Finally, all conservation efforts should be underpinned by education and community awareness schemes, aimed at improving ranchers' understanding of the endangered status of wild dogs, and potential ecological benefits conferred by their presence, so as to increase tolerance.

In conclusion, the key to improving the conservation status of wild dogs in South Africa lies in cooperating with landowners and local communities in developing strategies to encourage co-existence, using economic incentives.

7.3 The applicability of this study to the conservation of other carnivores

There is increasing awareness that a reliance upon parks and reserves is not sufficient to ensure adequate biodiversity conservation (Edwards & Abavardi 1998). The low densities, and large area requirements of large carnivores in particular, make it imperative that land occurring outside protected areas is included in conservation planning. By virtue of their life history characteristics however, large carnivores are difficult to conserve (Linnell et al. 2001), and these problems are magnified outside protected areas. The diet of many large carnivore species overlaps significantly with that of humans, predisposing them to conflict (Macdonald & Sillero-Zubiri 2002). Many large carnivore species range widely, and those occurring outside protected areas are likely to encounter a mosaic of land uses, covering areas in which they may conflict with humans in varying ways and to varying degrees, and areas in which their presence does not result in conflict. Consequently, effective conservation strategies for large carnivores outside protected areas are likely to be diverse and adaptive, and involve sociological and economic considerations, on top of ecological concerns. My study used a series of methods to identify the most appropriate, and most cost effective strategies for conserving wild dogs under varying conditions. A similar series of methods, moulded to specific circumstances, has general applicability in the design and implementation of conservation efforts involving large carnivore species, worldwide.

7.3.1 Ecological approaches

Determining the distribution and status of populations, and assessing the conditions under which populations are persisting represents an important first step towards conserving a species. Populations of carnivores occurring outside of protected areas are unlikely to receive the attention given to many species within parks and reserves in terms of monitoring, and numbers are likely to be little known, or potentially exaggerated where a species' presence causes conflict, and under estimated in the case of cryptic or nocturnal species. Consequently, population size and geographic distribution assessments are especially important for carnivore species occurring outside protected areas. The geographic area to be sampled, and the population size, spatial organisation, behaviour and life history of the species in question will dictate the most appropriate census technique (Wilson & Delahay 2001). The sighting data collection methods and analyses used in my study represent a cheap and effective way in which the distribution and status of large, rare, charismatic carnivores can be gauged over large areas, although the accuracy of the methods has not been tested. The methods have particular applicability to developing countries, with limited resources to respond to declines in carnivore populations. Obvious candidate species for the methods in the developing world include cheetahs *Acinonyx jubatus* in Africa (Gros 1998; Gros & Rejmanek 1999) and Asia, tigers *Panthera tigris* in Asia, and wolves *Canis lupus* in eastern Europe and Asia.

Designing conservation strategies requires that the area required for the conservation of a viable population, or of a minimum demographic unit of a species, is determined with allowance for natural or artificial dispersal of individuals (Smallwood 2001). Chapter 3 in my study outlined simple methods for estimating the minimum area required for the minimum demographic unit, or a viable population of any large carnivore species. The methods employed in Chapter 3 are simple, and easily employed by scientists and managers, given adequate data on prey population sizes and likely predator diet, increasing the applicability of the methods for developing countries. The most obvious application of these methods is in the planning stages of carnivore reintroduction programmes, a conservation management tool now being employed worldwide (Fischer & Lindenmayer 2000; Breitenmoser et al. 2001).

7.3.2 Sociological approaches

An obvious step in implementing a conservation programme involving large carnivores outside protected areas is to identify the reasons behind the initial population decline. Conflict with humans represents the most common cause of conservation problems for many large carnivores (Woodroffe & Ginsberg 1998) and the survey techniques employed in this study were adequate to determine if conflict exists between local people and carnivores, and why. Such techniques enable the design of strategies that might be employed to reduce or remove conflict, and permit identification of the conditions under which conflict is likely to be least prevalent. These methods are widely applicable to conservation planning for large carnivore species, and have been intensively used - for example prior to and during the expansion of wolf populations in Europe and the USA through reintroduction and natural re-colonisation (Lohr et al. 1996; Pate et al. 1996; Ericsson & Herbelein 2003).

7.3.3 Economic approaches

Conflict between large carnivores and humans frequently occurs as a result of perceived or real economic costs resulting from the loss of livestock or game animals. Consequently, conservation solutions based upon reversing or minimising economic losses due to the presence of large carnivores are likely to be effective in reducing the state of conflict. Ecotourism is a rapidly growing industry (Gossling 1999) and the derivation of economic benefits from wildlife and protected areas is a potentially important way in which conservation efforts can be justified (Norton-Griffiths 1995). Carnivores are especially popular with tourists, and significant potential exists in terms of deriving economic benefits from ecotourism, particularly from charismatic, highly visible species such as tigers, lions *Panthera leo* and wolves (Sillero-Zubiri & Laurenson 2001). The methods outlined in Chapter 5 permit rapid assessment of the potential tourism revenue from a carnivore species, relative to the costs of conservation programmes, and are easily applied to other species. These techniques are useful in identifying the conditions under which ecotourism benefits are likely to exceed the costs, and the conditions whereby ecotourism benefits are unlikely to match the costs and where conservation efforts are likely to depend upon donor funding.

In southern Africa, large areas of land are devoted to game ranching, with income based upon tourist hunting, or ecotourism (Earnshaw & Emerton 2000; van der Waal & Dekker 2000). This land tenure background is ideal for conservation planning based upon the exploitation of the ecotourism value of large carnivores such as cheetahs, leopards *Panthera pardus*, lions, and wild dogs. In Europe and the USA, tourists do not typically visit private ranches, and resultantly, conservationists have largely ignored the potential for exploiting the tourist value of large carnivores outside of state protected areas. The results of my study suggest that tourists are willing to pay to view a large carnivore in its natural habitat, and that the revenue is potentially sufficient to offset the costs of predation in most scenarios on private land. Expanding populations of brown bears *Ursus arctos*, lynx *Lynx lynx*, and wolves in Europe, and cougars *Felis concolor* and wolves in North America are increasingly conflicting with private land owners as a result of increasing livestock depredations, and competition with humans for wild prey (Breitenmoser 1998; Mech 1998). I suggest that European and North American conservationists might consider establishing ecotourism operations on private land as a means of creating economic incentives for land owners to conserve large carnivores occurring on their properties. Similar schemes might be considered in the cattle ranching regions of the Pantanal in Brazil, to reduce persecution of jaguars *Panthera onca* and cougars (Johnson et al. 2001). Likewise, ecotourism schemes have the potential to promote coexistence between people and large carnivores such as cheetahs, lions, wild dogs in east Africa, and tigers in parts of Asia, where land tenure systems are based more upon communal ownership and subsistence farming (Cumming 1991; Damania et al. 2003).

Increasing numbers of threatened and endangered species are resulting in an increasing shortfall between the resources required to ensure effective conservation, and the resources that are actually available (Myers et al. 2000). Consequently, conservation programmes must deliver value for money. The estimation and comparison of costs of proposed conservation strategies represents a key step in programme design. The cost efficiency indices developed in this study to compare the long term costs of competing conservation strategies provide a solid basis from which to decide between potentially competing conservation strategies for any threatened or endangered species, with particular applicability to species such as tigers and wolves which attract large sums of donor funding (Mech 1998; Christie 2001).

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APPENDIX A

Appendix A Newspapers, newsletters and magazines in which appeals for wild dog sightings were published

Publication	Type	Language	Circulation of publication
Africa Environment and Wildlife	Magazine	English	National
Farmers Weekly	Magazine	English	National
Game and Hunt	Magazine	English	National
The Natal Courier	Newspaper	English	KZN [a]
The Vryheid Herald	Newspaper	English	KZN
The Zululand Observer	Newspaper	English	KZN
Die Soutpansberger	Newspaper	Afrikaans	LP [b]
The Letaba Herald	Newspaper	English	LP
The Northern Post	Newspaper	English	LP
The Phalaborwa Herald	Newspaper	English	LP
The Phalagraph	Newspaper	English	LP
The Waterberg Newsletter	Newsletter	English	LP
The Lowvelder	Newspaper	English	LP, MP [c]
Transvaal Game Association	Newsletter	English	LP, MP, NW [d]
Die Bosvelder	Newspaper	English	LP, NW
The Lydenburg News	Newspaper	English	MP
The Middelburg Observer	Newspaper	English	MP
Die Noordwester	Newspaper	English	NW
Die Stellalander	Newspaper	Afrikaans	NW, NC [e]

[a] Kwa Zulu Natal
[b] Limpopo
[c] Mpumalanga
[d] North West
[e] Northern Cape

APPENDIX B

Appendix B Questionnaire used to assess the attitudes of ranchers

Dear Sir/Madam,

This questionnaire survey is being conducted as part of a wider study that aims to determine the costs and benefits associated with the conservation of African wild dogs *(Lycaon pictus)*. This survey aims to determine the attitudes of you as landowners towards this species on private land, and the reasons behind these attitudes. You assistance in completing this questionnaire survey would be greatly appreciated. Your answers will be anonymous and entirely confidential.

Section 1. Ranch Details

1. What is your position?

- ☐ Land owner
- ☐ Lease holder
- ☐ Manager
- ☐ Other, please specify

2. Please provide the following ranch details

Ranch name...................................

Province.......................................

District.......................................

Nearest town...................................

Distance to nearest town.....................

3. What is the size of the property in hectares/acres?

4. Is your property part of a collaborative nature reserve (CNR) or conservancy?

5. If so, what is the size of the CNR or conservancy?

6. If the ranch is not part of a CNR, please provide the following details concerning the fencing on the property

Location of fencing...

Height of fence...

Number of strands...

Is the fence meshed (Bonnox / Veldspan)?..............................

Is the fence electrified?..

7. What type of land borders your property?

- ☐ State land
- ☐ Communal land
- ☐ Forest land
- ☐ Cattle ranch
- ☐ Mixed cattle and game ranch
- ☐ Game ranch
- ☐ Other, please specify

8. Please indicate the relative importance of the following sources of income on your property

Activity	Important	Marginal	Zero
Trophy hunting			
Hunting lease rights			
Ecotourism			
Live Game Sales			
Venison Sales			
Cattle			
Sheep			
Crops			

9. Please indicate the severity of poaching on your property

(0=no poaching, 5=severe poaching)

0☐ 1☐ 2☐ 3☐ 4☐ 5☐

10. If poaching occurs on your property, by what means does it occur?

Poaching method	% of total poaching
Snaring	
Hunting with dogs	
Shooting	
Other, specify...........................	

Section 2. Predators

11. Which of the of following predators are regularly sighted on your property?

- ☐ Brown hyaena
- ☐ Caracal
- ☐ Cheetah
- ☐ Feral dogs
- ☐ Jackals
- ☐ Leopard
- ☐ Lion
- ☐ Spotted hyaena
- ☐ Wild dogs
- ☐ Others, please specify......................................

12.Indicate how you feel about having (or how you would feel about having) each of the following species on your land by giving each species a score of 0-5 (0=Very negative, 5=Very positive).

Species	Score	Reason
African wild dogs		
Cheetahs		
Jackals		
Lions		
Leopards		
Spotted hyaenas		

Section 3. Wild Dogs

13. How frequently is evidence of the presence of wild dogs seen on your property (including sightings, spoor, kills etc.)

14. If wild dogs occur on your land, how many do you estimate use the property?

15.Have wild dogs had pups on your land in the last five years?

16. If wild dogs occur on your land, what species do they prey upon most regularly?

a)……………………

b)…………………..

c)……………………

17. Given the choice between having wild dogs utilise your property as part of a larger home range, or not having them at all, which would you choose?

☐ Wild dogs utilising your property as part of their home range

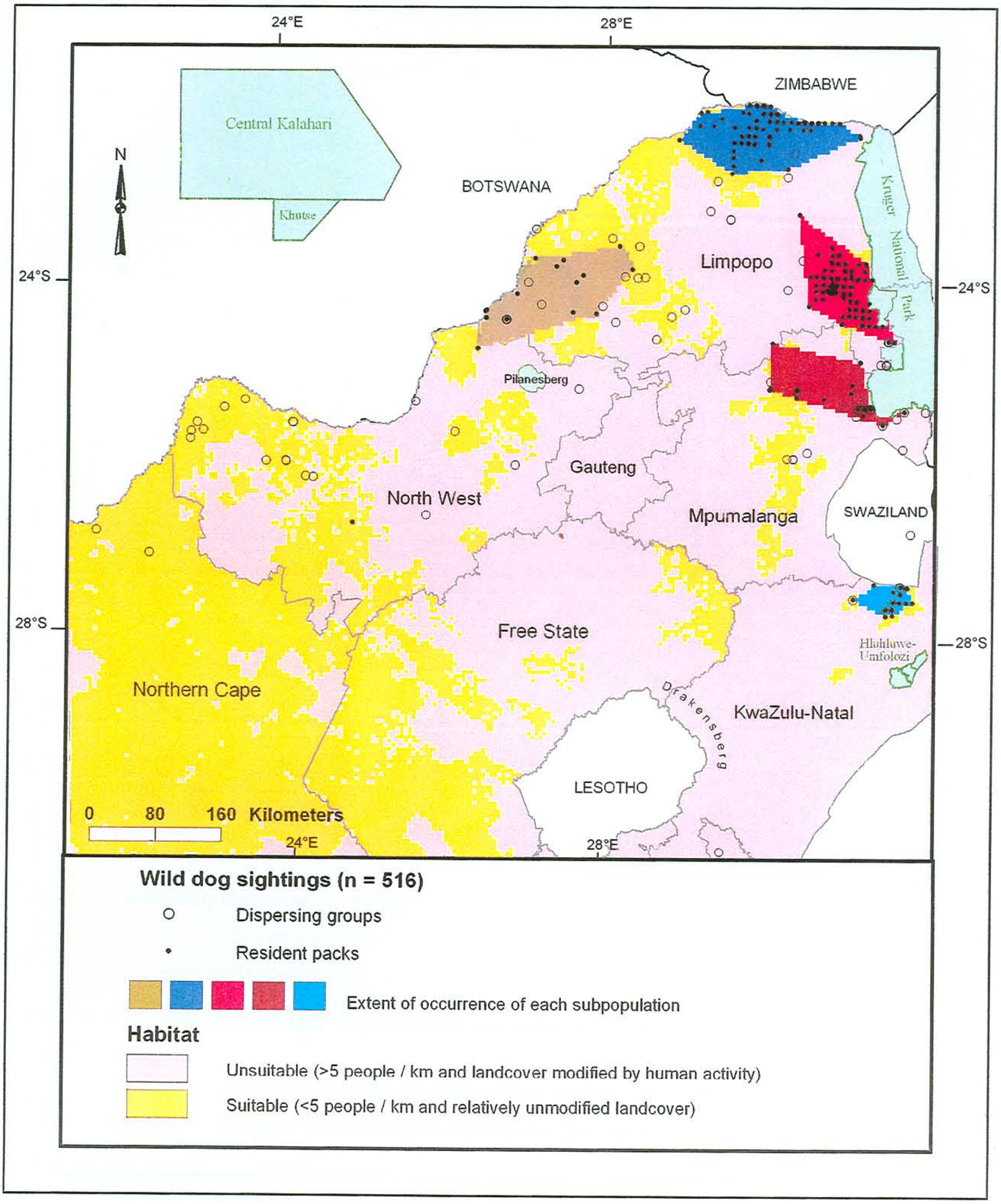
☐ Wild dogs not utilising your property at all

18. Why?

19. Please indicate whether you agree with the following statements

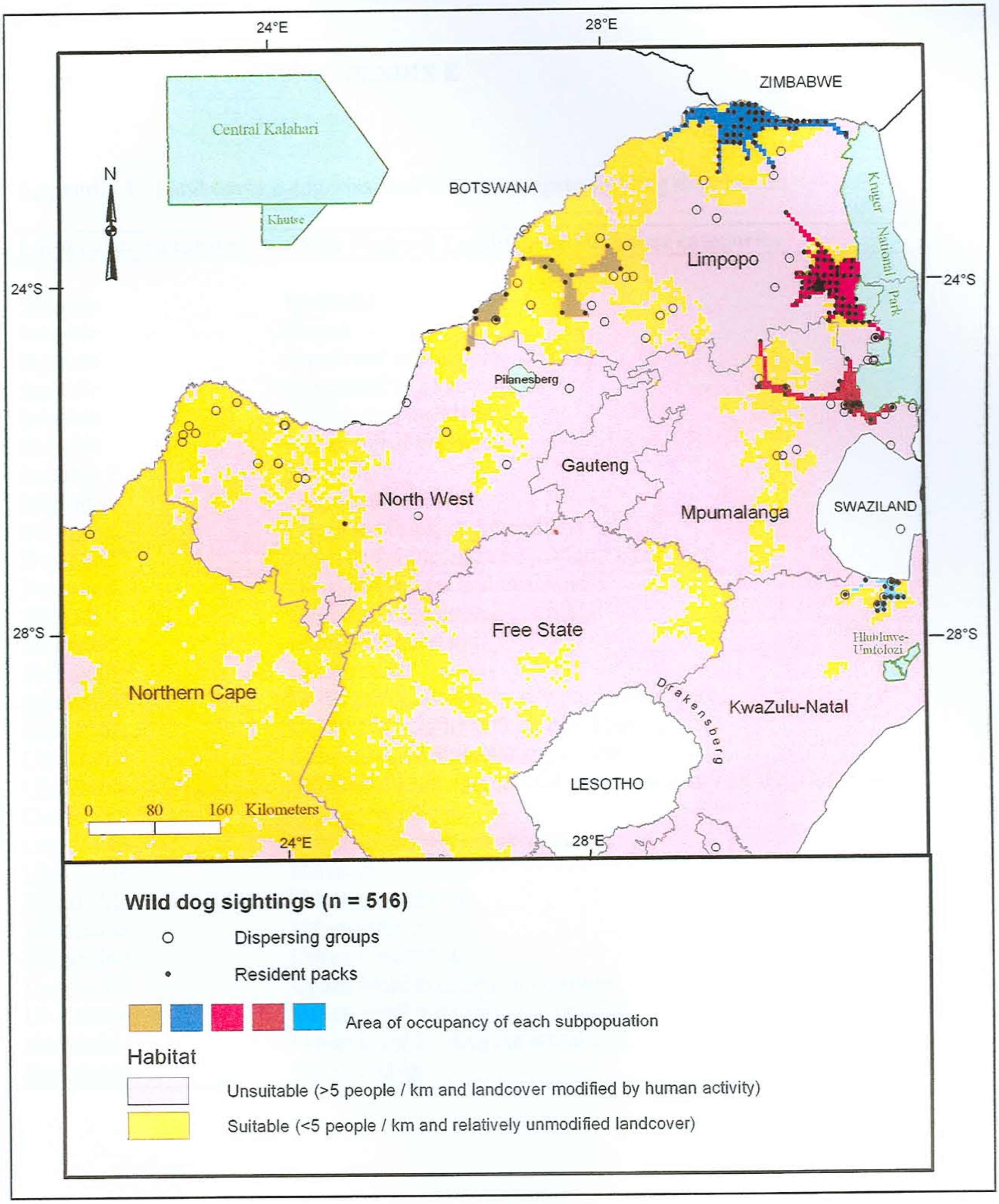
Statement	Yes	No	Don't know	Comments
A. Wild dogs are a liability to a game farmer because they consume valuable game but provide no economic return.				
B. Wild dogs regularly kill more food than they need				
C. Wild dogs cause disruption of game herds and make them more skittish				
D. Wild dogs cause damage to fences during their hunting activities				
E. Wild dogs are a natural component of a healthy ecosystem				
F. Tourists are interested in seeing wild dogs				
G. Sufficient money can be made from marketing 'wild dog eco-tours' to compensate for the losses caused by their predation				
H. The demonstration of a way in which to make a sustainable profit from wild dogs would be sufficient incentive to encourage you to reintroduce wild dogs onto your property				

APPENDIX C



Appendix C. Maximum area estimate of the geographic range of resident subpopulations of wild dogs outside protected areas in South Africa

APPENDIX D



Appendix D. Minimum area estimate of the geographic range of resident subpopulations of wild dogs outside protected areas in South Africa

APPENDIX E

Appendix E Land cover categories used in to investigate sighting distribution

Land cover suitability	CSIR National Land Cover Database categories
Suitable	Herbland
Suitable	Forest
Suitable	Forest and woodland
Suitable	Shrubland and low Fynbos
Suitable	Thicket and bushland
Suitable	Unimproved grassland
Suitable	Wetlands
Degraded	Degraded herbland
Degraded	Degraded forest and woodland
Degraded	Degraded shrubland and lowland Fynbos
Degraded	Degraded thicket and bushland
Degraded	Degraded unimproved grassland
Degraded	Improved grassland
Unsuitable	Barren rock
Unsuitable	Cultivated, commercial dry land agriculture
Unsuitable	Cultivated commercial irrigated agriculture
Unsuitable	Cultivated, commercial sugarcane
Unsuitable	Cultivated semi commercial subsistence dry land agriculture
Unsuitable	Forest plantations
Unsuitable	Gullies and sheet erosion scars
Unsuitable	Mines and quarries
Unsuitable	Urban commercial
Unsuitable	Urban industrial
Unsuitable	Urban residential
Unsuitable	Urban small holdings in bushland
Unsuitable	Urban small holdings in grassland
Unsuitable	Urban small holdings in woodland
Unsuitable	Water bodies

APPENDIX F

Appendix F The source of cost estimates for the reintroduction of wild dogs into private reserves, and for ecotourism schemes (addresses available from the author)

Reference number	Organisation / company
1	Rhino Fencing
2	Harrop Allen Fencing
3	Automobile Association of South Africa
4	Moller & Moller Guns & Sport
5	A survey of professional game-capture and sale teams (n=10)
6	De Wildt Cheetah Breeding Research Centre
7	Kruger National Park
8	Wildlife Translocation Society of South Africa
9	South African Veterinary Association
10	ANB Veterinary Suppliers
11	Dan inject International (South Africa)
12	Wildlife Broking Services
13	McCarthy Used Car Dealership
14	Biotrack
15	Sirtrack
16	Wildlife Materials
17	University of Pretoria
18	Field Guides Association of South Africa
19	Getaway Magazine

APPENDIX G

Appendix G Questionnaire used to assess tourist's willingness to pay to see wild dogs

African Wild Dog Conservation Tourist Questionnaire

The African wild dog (also known as the painted hunting dog) is a highly social wild carnivore, occurring in tight-knit packs of between 2-40. They have large round ears, blotched black, tan and white coats and long legs. African wild dogs were formerly very widespread across most of Africa. However, they have suffered a major reduction in their range and now occur in appreciable numbers in only 6 of the 39 countries in which they once occurred. As a result of this, the African wild dog is a highly endangered species.
There are large areas of suitable habitat available for the reintroduction of African wild dogs in South Africa. However, wild dogs are unpopular with ranchers because they are perceived to be very expensive to keep, whilst yielding little or no financial return. Until a value is identified for wild dogs, the conservation status of the species in this country is unlikely to improve. In this questionnaire, we aim to see how interested you, as tourists, are in seeing wild dogs.

SECTION 1. HOLIDAY DETAILS

This section is aimed at obtaining a description of the type of holiday that you are on.

1. What is the duration of your stay at Ngala?

- ☐ Day Visitor
- ☐ 1-3 nights
- ☐ 4-5 nights
- ☐ 6-7 nights
- ☐ 8-10 nights
- ☐ 11-14 nights
- ☐ 15+ nights

2. How many days have you spent at Ngala on this trip so far?

- ☐ First day
- ☐ 1-3 nights
- ☐ 4-5 nights
- ☐ 6-7 nights
- ☐ 8-10 nights
- ☐ 11-14 nights
- ☐ 15+ nights

3. What type of accommodation are you staying in?

Accommodation type	
Hotel	☐
Chalets	☐
Other, please specify..	

4. Please indicate the size of your family/group on this trip in the box below

Number of adults	
N umber of children (under 16)	

5. Please estimate the costs of your visit to Ngala for your family/group in the table below. You can use any currency, but please specify the currency of your answer.

Cost type	Cost	Currency
Airfare		
Ground transportation		
Park entry		
Accommodation		
Miscellaneous		

6. How many trips have you made to African Game parks/protected areas in the last five years, prior to this visit?

SECTION 2. VIEWING PREFERENCE

This section is aimed at determining your viewing preferences

7. Please indicate which of the following species you have seen on this trip with a tick.

Species	Yes	No	Not sure
African wild dog	☐	☐	☐
Black backed jackal	☐	☐	☐
Cheetah	☐	☐	☐
Leopard	☐	☐	☐
Lion	☐	☐	☐
Spotted hyaena	☐	☐	☐

8. Please give each of the species listed in the table below a score of 0-5 in terms of the desire you have to see each on this trip (0=not at all, 5=very much so) by ticking the relevant box. If there are species not listed below that you would very much like to see, please include them under the 'other' boxes below and give them a score.

Species	0	1	2	3	4	5
African wild dog	☐	☐	☐	☐	☐	☐
Buffalo	☐	☐	☐	☐	☐	☐
Cheetah	☐	☐	☐	☐	☐	☐
Elephant	☐	☐	☐	☐	☐	☐
Giraffe	☐	☐	☐	☐	☐	☐
Hippopotamus	☐	☐	☐	☐	☐	☐
Impala	☐	☐	☐	☐	☐	☐
Kudu	☐	☐	☐	☐	☐	☐
Leopard	☐	☐	☐	☐	☐	☐
Lion	☐	☐	☐	☐	☐	☐
Rhino	☐	☐	☐	☐	☐	☐
Sable	☐	☐	☐	☐	☐	☐
Spotted hyaena	☐	☐	☐	☐	☐	☐
Wildebeest	☐	☐	☐	☐	☐	☐
Zebra	☐	☐	☐	☐	☐	☐
Other.....................	☐	☐	☐	☐	☐	☐
Other.....................	☐	☐	☐	☐	☐	☐
Other.....................	☐	☐	☐	☐	☐	☐

9. Please give each of the following reserve characteristics a score of 0-5 in terms of how important they are to you on a trip to a wildlife area by ticking the relevant box (0=of no importance, 5=highly important).

	0	1	2	3	4	5
A. The presence of attractive scenery						
B. The presence of a high bird diversity						
C. The presence of a high mammal diversity						
D. The presence of a high floral diversity						
E. The presence of the 'big 5' species						
F. The presence of large predators						
G. The presence of wild dogs						

10. Did the presence of wild dogs at Ngala influence your decision to visit the park?

☐ Yes
☐ No

11. Have you ever seen wild dogs in the wild on previous trips to protected areas?

☐ Yes
☐ No

12. Prior to completing this questionnaire, were you aware that African wild dogs are endangered?

- ☐ Yes
- ☐ No

13. Although wild dogs are present at Ngala, you have less than 20% chance of seeing them during your stay. During three months in winter, wild dogs remain in the vicinity of a den in which they have pups. During this time, the location of African wild dogs is very predictable and guided trips to see them would be able to almost guarantee sightings. How much would you pay per person to go on an optional, small (6 person max) guided tour to a wild dog den, in order to improve your chance of seeing wild dogs to more than 90%? Such a trip would be conducted in a fashion designed to result in minimal disturbance to the African wild dogs. You can use any currency, but please specify the currency of your answer.

Amount...

Currency...

14. If you answered 0 for this question, please explain why below

SECTION 3. PERSONAL DETAILS

Finally, in this section, we would like to find out a little about you so that we can see how different people feel about the issues we are investigating

15. Gender:
- ☐ Male
- ☐ Female

16. Age
- ☐ 10-20
- ☐ 20-30
- ☐ 30-40
- ☐ 40-50
- ☐ 50-60
- ☐ 60+

17. What is the current level of education that you have experienced?

- ☐ Didn't complete school
- ☐ Secondary school
- ☐ Tertiary (degree, diploma)
- ☐ Higher degree

18. What is your country of permanent residence?

19. If you are southern African, please state your town or city and province

Town/City........................

Province...........................

Thank you very much for completing this questionnaire. Your contribution is important in enabling the completion of this research project and ultimately will be of benefit in the conservation of the wild dog.

AFRICA

CCAFRICA SAFARI DESTINATIONS

CCAfrica supports African wild dog conservation. We would like to thank you for completing this questionnaire and contributing information towards African wild dog conservation-research. We would like to make it clear that the questions posed in this survey are of a purely hypothetical nature. CCAfrica does not and will not charge extra for African wild dog viewing opportunities.

APPENDIX H

APPENDIX H The breakdown of expenditure upon the South African wild dog meta-population during 1996 – 2001 (ZAR in parentheses)

	Hluhluwe	Karongwe	Madikwe	Pilanesberg	Venetia
Initial reintroduction					
Fencing upgrade				8,889 (97,694)	26,055 (286,348)
Holding facilities upgrade		1,442 (15,848)		2,785 (30,604)	3,410 (37,475)
Community outreach				4,601 (50,561)	5,099 (56,037)
Purchase / provision of dogs		1,456 (16,000)			2,184 (24,000)
Capture and care		865 (9,511)		353 (3,883)	3,810 (41,875)
Transport of dogs		1,304 (14,328)		3,427 (37,664)	1,043 (11,463)
Feeding in holding facility				4,757 (52,283)	5,984 (65,760)
Monitoring / research [a]		305 (3,351)			28,463 (312,804)
Other				706 (7,758)	3,702 (40,691)
Maintenance					
Holding facility upgrade	11,539 (126,812)				
Community outreach	722 (7,936)				
Purchase / provision of dogs			9,876 (108,541)		
Capture and veterinary care	1,250 (13,737)		6,818 (74,934)		
Transport of dogs	898 (9,870)		882 (9,689)	184 (2,020)	
Feeding in holding facility	981 (10,777)		1,319 (14,498)	3,046 (33,472)	
Monitoring / research [a]	41,687 (458,144)	23,923 (262,914)	21,144 (232,368)	19,173 (210,712)	
Consultancy			519 (5,703)		
Total	57,077 (627,276)	29,295 (321,952)	40,558 (445,733)	47,921 (526,651)	79,750 (876,453)

[a] Including labour and mileage expenses incurred as a result of personnel attending Wild dog Advisory Group-South Africa meetings