The sustainability of leopard *Panthera pardus* sport hunting in Niassa National Reserve, Mozambique

By

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As the candidate's supervisor, I have approved this thesis for submission.

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ABSTRACT

Leopard *Panthera pardus* are an economically valuable asset and when used in sustainable consumptive use programs can provide tangible benefits to communities to improve human livelihoods and the conservation of the species. Sport hunting is increasingly proposed as a tool to generate funds to support the conservation of leopard and other large carnivores. However, to assess the value of sport hunting as a conservation tool it is critical to understand its economic impact and ensure that the off-takes are sustainable. In this study I assessed the conservation status of leopard and the ecological sustainability of legal and illegal off-take in Niassa National Reserve (NNR) the largest protected area, 42,000 km^2, in Mozambique, which is inhabited by 35,000 people. I also investigated whether the revenues from leopard sport hunting off-set the costs of depredation on livestock in local communities and individual benefits from poaching by local hunters. To perform this study, I interviewed hunting operators and villagers, collected camera trapping data, and analyzed long-term leopard sport hunting data. Leopard had high value for sport hunters, however, the economic benefits from the legal hunting did not off-set the costs from livestock depredation and did not compete with benefits from the illegal hunting which accrued to individuals at the household level. Leopard population densities in Niassa Reserve were comparable with the study sites in central and southern Africa. The numbers of leopard legally hunted in NNR appear to be ecologically sustainable, however a high percentage of the leopard taken as trophies were under the recommended age of seven years. The illegal off-take was unsustainable and resulting in high turnover and combined with the operators' off-take is likely to be negatively affecting leopard populations. For the future ecological and economic sustainability of leopard quotas, I recommend improvements in the distribution of economic benefits and creating economic incentives to encourage villagers not to engage in the illegal hunting and quantification and inclusion of the illegal off-take in the annual quotas. My study also indicates the need to zone community and wildlife areas in NNR to reduce the anthropogenic effects on leopard and other carnivore populations.

PREFACE

The work described in this dissertation was carried out in the Niassa National Reserve, Mozambique, through the School of Life Sciences, University of KwaZulu-Natal, Durban, under the supervision of Prof. Rob Slotow and co-supervision of Drs Abi Vanak and Colleen Begg.

These studies represent original work by the author and have not otherwise been submitted in any form for any degree or diploma to any tertiary institution. Where use has been made of the work of others, it is duly acknowledged in the text.

Agostinho A. Jorge **Date**

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DECLARATION 1 - PLAGIARISM

I, Agostinho Aquelino Jorge, declare that

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DECLARATION 2 - PUBLICATIONS

Details of contribution to publications that form part and/or include research presented in this thesis (include publications in preparation and submitted and give details of the contributions of each author to the experimental work and writing of each publication).

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AJ conducted the field work, processed and analyzed the data, and wrote the paper. CB provided sport hunting and human-leopard conflict data. AV, MT, CB and RS provided valuable inputs for the manuscript.

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CHAPTER 1

General Introduction

Large carnivores are among the most difficult taxa to conserve because they are predisposed to conflict with people (Treves 2009). As the result of human actions some large carnivore species have become extinct (Falklands wolf *Dusicyon australis*, Gittleman & Gompper 2001) while others (lion *Panthera leo,* Henschel *et al.* 2010; tiger *P. tigris,* Dinerstein *et al.* 2007; cheetah *Acinonyx jubatus*, Marker & Dickman 2004; and jaguar *P. onca*, Sanderson *et al.* 2002) are experiencing dramatic declines in their range. Although there are increases in the numbers of some carnivore populations as a result of more efficient management in parts of southern Africa (Hayward *et al.* 2007), North America (Linnell *et al.* 2001) and Asia (Singh & Gibson 2011), the rapid expansion of human populations and associated unregulated hunting (Woodroffe 2000, Datta *et al.* 2008, Velho *et al.* 2012) in most parts of the world might erode the present conservation successes if appropriate measures are not taken.

1.1. Conservation of large carnivores

1.1.1 Importance of conserving large carnivores

Large carnivores are charismatic animals, valued by humans for ecological, cultural, aesthetic, spiritual and economic reasons (Karanth & Chellam 2009). They play an important role in maintaining healthy ecosystems, by controlling numbers of other predators through competition, and influencing populations of prey species (Miller *et al.* 2001). Efforts to protect ‘umbrella’ species such as large carnivores can also have positive effects on the conservation of other wildlife species (Linnell *et al.* 2000, Caro 2003). Thus, the removal of large predators from their natural habitat can have adverse impacts on community structure, which might negatively impact on the local ecology (Miller *et al.* 2001). Large predators are the main attraction in the consumptive and non-consumptive tourism industry in Africa (Macdonald & Sillero-Zubiri 2002). People also admire and revere carnivores because of their spiritual importance in many human societies (Hobgood-Oster 2010).

1.1.2 Management and monitoring of large carnivores

Large carnivores occur in a variety of ecological and social settings, and require different strategies for their effective conservation (Karanth & Chellam 2009). They often occur at low population densities because of their high position in the food chain (Skinner & Chimimba 2005). Large carnivores have large home ranges, and require vast areas for conservation of viable populations (Linnell *et al.* 2000, Macdonald & Sillero-Zubiri 2002). Many carnivores are also nocturnal, solitary, elusive and occasionally dangerous, making them difficult to study (Karanth *et al.* 2010a).

Monitoring of carnivore population size and trends is one of the most important components of their management, particularly for populations subject to harvest programs (Linnell *et al.* 1998, Harrison 2006), but it is often practically and technically challenging to obtain reliable estimates (Funston *et al.* 2010). Techniques to monitor terrestrial carnivores are diverse (Gese 2001, Wilson & Delahay 2001, Karanth *et al.* 2010a), and include direct methods such as mark-recapture (Gese 2001) as well as indirect methods such as track transects, DNA analysis (Dallas *et al.* 2003), and individual voice recognition (Terry & McGregor 2002). With the development of new techniques such as camera trapping and GPS collars, researchers improved their ability to studying carnivores (Karanth *et al.* 2010a). These advancements provide more robust data, allowing evidence based inputs to assist wildlife authorities with the sustainable management of large carnivore populations (Wegge *et al.* 2004), such as with harvesting policies of infanticidal species including lion and leopard *P. pardus* (Packer *et al.* 2009, Balme *et al.* 2010). However, to make carnivore conservation work, the biological data obtained through various methods described should be complemented with socio-economic and political information (Weber & Rabinowitz 1996).

1.1.3 Threats to large carnivore conservation

Conflict with humans, habitat loss and fragmentation, and human hunting are among the most serious threats to large predators' conservation (Ray *et al.* 2005). Human-carnivore conflict is a major conservation and rural livelihood concern, and can have significant impacts on both human and wildlife populations (Dickman 2008). The problem is severe where human expansion and changes in land use have increased the competition between people and large carnivores, especially around protected areas (Chardonet *et al.* 2010). The continued human demographic pressure, and the associated demand for food and natural resources, will reduce and fragment further remaining carnivore habitats (Karanth & Chellam 2009). Humans also threaten carnivores through snaring (Chardonet *et al.* 2010) both directly by inadvertent snaring of the carnivores in snares set for bushmeat (Loveridge *et al.* 2007), and indirectly through removal of prey species (Bennett *et al.* 2006, Secretariat of the Convention on Biological Diversity 2011). Snaring has higher detrimental impacts on carnivores living close to human settlements (Loveridge *et al.* 2010), and affects the larger carnivores more than smaller ones (Carbone *et al.* 2011). Because of the strong relationship between high human population densities and carnivore declines (Woodroffe 2000), conservationists need to distinguish areas exclusively for carnivore conservation, areas where humans and predators coexist and areas where predators are not tolerated (Emerton 2001).

1.2. Approaches to conserve large carnivores

1.2.1. Importance of protected areas for carnivore conservation

Protected areas saved many large carnivore species from extinction in the past (Woodroffe & Ginsberg 1998) but they are not the final solution for carnivore conservation (Weber & Rabinowitz 1996). Some carnivore species', including African wild dog *Lycaon pictus* and cheetah, occur at low densities in protected areas because of intra-guild competition with larger carnivores (Caro & Stoner 2003). However, in communal areas these species occur at relatively high densities thus these areas are critical to conserve the species (Marker *et al.* 2003, Schumann *et al.* 2008). For large carnivores such as lion, protected areas are still a relevant approach for their conservation (Croes *et al.* 2011). Nonetheless, in some protected areas the potential to conserve wildlife is challenged by the presence of human communities living inside them (Adams & Hutton 2007). In addition, there are few protected areas large enough to conserve viable populations of large carnivores (Forbes & Theberge 1996, Woodroffe & Ginsberg 1998, Cantu-Salazar & Gaston 2010) and this often results in carnivores dispersing and establishing subpopulations outside protected areas (Singh & Gibson 2011). Thus, for effective conservation of predators, a balance between protected areas and communal or private land seem more appropriate, with large areas allocated to conservation through multiple-use management systems (Van Aarde & Jackson 2007, Redford *et al.* 2011).

1.2.2. Economic value of conserving large carnivores

Carnivores are economically valuable assets (Loveridge *et al.* 2006), and their sustainable use through photographic tourism or sport hunting can contribute to both human livelihoods and conservation of their habitats (Loveridge *et al.* 2010). In some instances hunting accounts for a small proportion of mortality compared with other human mediated causes (Balme *et al.* 2010) and can be sustainable in the long-term (Dowsley 2008, Nilsen *et al.* 2011). For example, hunting of Canadian lynx *Lynx canadensis* finances the species' long-term protection (Nowell & Jackson 1996). The key for the success of Canada lynx has been clear policies, effectively applied legislation, and population monitoring and research (Loveridge *et al.* 2010).

Different proportions of sport hunting revenues accrue to various stakeholders in the hunting industry (Mayaka *et al.* 2005). In most of southern Africa countries, private companies and state agencies capture the majority of hunting profits (Nelson 2009), while a relatively small proportion accrues to local communities, the custodians of the resource (Baker 1997, Lewis & Alpert 1997, Lindsey *et al.* 2007a). Furthermore, the revenues distributed often are not translated into positive community actions towards wildlife because benefits are not conditioned or linked to the desired conservation actions (Sachedina & Nelson 2010).

Sport hunting is a highly contentious issue within environmental spheres with both proponents and opponents of the activity (Badenhorst 2003), especially because it involves the killing of animals in situations where it is not necessary for the survival of individual hunters (Pacelle 1993, Gorski 2011). In addition, some species whose populations are in continued decline are still hunted (Croes *et al.* 2011). There are also concerns regarding the dangers of ethical abuses because sport hunting is driven by open market rules and substantial revenues are involved (Lindsey *et al.* 2007a, Loveridge *et al*. 2010). Patterson (1998) described canned hunting in South Africa as a notable example of abuses in the hunting industry. Furthermore, if poorly regulated, carnivore hunting can cause population decline due to excessive quotas (Spong *et al.* 2000, Caro *et al.* 2009), or induce behavioral effects such as increased levels of infanticide and long-term genetic change (Whitman *et al.* 2004, Loveridge *et al.* 2007, Packer *et al.* 2009, Croes *et al.* 2011). Also, to be sustainable and competitive, land managed for sport hunting should include anti-poaching and community development activities (Barnett & Patterson 2006). Therefore, care needs to be taken to ensure its sustainability through careful monitoring of quotas (Leader-Williams 2009). The question as to whether leopard sport hunting can be used as a conservation tool is a good example of the social, economic, political, and ecological arguments relevant to biodiversity conservation (Badenhorst 2003, Leader-Williams 2009).

Another potential economic use of large carnivores is through photographic tourism, which generates significant sums for their conservation in developing countries (Gossling 1999). For example, in South Africa costs of maintaining African wild dog have been covered with revenues generated by tourists viewing dogs (Lindsey *et al.* 2005a). However, it is recognized that photographic tourism by itself, even if well planned, has a limited ability to solve all financial problems facing conservation and thus should not be seen as a panacea (Gossling 1999, Banerjee

2010). In addition, too high visitor pressure in tourism destinations can be ecologically unsustainable for the environment, even if direct removal of species does not occur (Baldus 2003, Meletis & Campbell 2007). The photographic tourism demands for aesthetic areas (Novelli *et al.* 2006), with associated tourist consumption of goods and services can generate environmental problems in destinations when there is a lack of infrastructure or facilities to deal with waste and sewage management (Meletis & Campbell 2007).

1.3. Local communities and conservation of large carnivores

1.3.1. Human-carnivore conflict

Large carnivores can cause significant economic losses and pose threats to human life (Holmern *et al.* 2007). The conflict is usually exacerbated by expansion of human activities into wildlife habitats, recovery by some carnivore populations and environmental changes (Treves 2008). This results in depredation of livestock and game populations, and occasional attacks on humans (Marker *et al.* 2003, Treves & Karanth 2003, Treves 2008, Inskip & Zimmermann 2008). The depredation on livestock is the most common source of conflict between humans and carnivores (Inskip & Zimmermann 2008). Carnivore attacks on livestock represent a significant problem for the subsistence of villagers because domestic animals represent not only a source of protein but they are also a symbol of wealth and savings (Chardonet *et al.* 2010). Although the proportions of livestock losses to carnivores vary with the size of the herds, husbandry techniques and density of predators (Chardonet *et al.* 2010, Loveridge *et al.* 2010), they are often smaller than losses caused by diseases, poor livestock husbandry or theft (Rasmussen 1999, Kissui 2008). Nonetheless, losses imposed by carnivores can be significant at the household level, and can motivate people to kill many individual carnivores in retaliation to the depredation caused by only a few individuals (Treves 2008). For instance, economic losses caused by large carnivores in Tanzania and Kenya (see Woodroffe *et al.* 2005, Nelson 2009) result in large numbers of lion, spotted hyena *Crocuta crocuta* and leopard being poisoned or speared (Dickman 2008, Kissui 2008). In Namibia, cheetah are persecuted by farmers because they kill economically valuable wild ungulates on commercial game farms (Marker *et al.* 2003).

While large carnivore attacks on humans are less common than on livestock and game species, they represent an important concern for people (Treves 2008), and can intensify the conflict, compromising efforts to conserve predators (Thirgood *et al.* 2005). The consequences of conflict are particularly bad when carnivores kill adult men, who are key income generators in rural households (Gurung *et al.* 2008).

Recently, to alleviate concerns related to the carnivore attacks on humans, Mozambican wildlife authorities, in collaboration with tourism operators, killed at least 51 crocodile *Crocodylus niloticus* and collected about 9,600 eggs along the Zambezi River (AllAfrica 2011). In Tanzania, attacks by large carnivores on humans are a growing concern, although they are restricted to few areas (Nyahongo & Roskaft 2011). There, lion killed at least 560 people and injured more than 308, with a remarkable increase in the annual rate of attacks since 1990 (Packer *et al.* 2005).

1.3.2 Mitigation of Human-carnivore conflict

Reducing human mediated mortality on carnivore populations is an important strategy to maintain predators where they coexist with people (Woodroffe 2000). To address economic losses caused by carnivores, government institutions and conservation organizations often pay to compensate individuals losing livestock (Lee 2011). Until recently, cash payments were made to cover economic losses caused by carnivores only after the depredation event had occurred (Mishra 1997). However, criticisms of the effectiveness of this approach (Bulte & Rondeau 2007), contributed to development of a new strategy, compensation in advance (Zabel & Holm-Müller 2008). Payment in advance is related to the concept of Payment for Ecosystem Services (PES) and aims to reconcile costs imposed by carnivores at local community level with their high values at national and international levels (Nelson 2009). In this strategy, payments are usually linked to a desired conservation success and are made based on the expectation that the damage is likely to occur (Nelson 2009, Loveridge *et al.* 2010). Payments have been implemented with different degrees of success on conservation of various carnivore species (wolverine *Gulo gulo* and Eurasian lynx *Lynx lynx* in Sweden, Zabel & Holm-Müller 2008; Jaguar in Mexico, Nistler 2007; and snow leopard *P. uncia* in India, Mishra *et al.* 2003).

1.3.3 Coexistence between people and large carnivores

People seem to accept living alongside dangerous animals when they own or benefit from them (Hutton & Leader-Williams 2003, Treves 2009), and the level of tolerance is related to the magnitude of the damage and level of education (Dickman 2008, Loveridge *et al.* 2010). Tourism revenues have the potential to increase peoples' willingness to coexist with predators (Oli *et al.* 1994, Romañach *et al.* 2007). Effective revenue distribution among community members motivated villagers to allow lion to return to communal lands in Namibia (NACSO 2008). The knowledge of people about large carnivores determines how they value the animals, therefore education of rural villagers plays a key role in conservation programs (Sillero-Zubiri *et al.* 2007).

Intolerance is often based on misconceptions about the real potential risk predators pose to livestock and people (Treves & Karanth 2003). For example, in Kenya, villagers' perception of lion, spotted hyena and leopard as the most serious predators of livestock caused less negative attitudes towards wild dog (Woodroffe *et al.* 2005). Conversely, in South Africa ranchers had more negative attitudes towards wild dog because they did not benefit from them, contrasting with leopard which generated benefits for tourism (Lindsey *et al.* 2005b). Erroneous or not, it is important to address peoples' perceptions regarding predators because people are what ultimately drives the removals of carnivores (Rasmussen 1999, Sillero-Zubiri & Laurenson 2001).

1.4. Leopard

1.4.1. Conservation status of leopard

Its adaptability and ability to live in diverse environmental conditions, and its tolerance of anthropogenic impacts, distinguish the leopard from many other large carnivores (Nowell & Jackson 1996). Leopard are found in much of sub-Saharan Africa and Asia, but small populations also occur in the Middle East and south-eastern Europe (Hunter *et al.* 2003). They are endangered where populations have become heavily fragmented and isolated, especially in the Middle East and northern Africa (Uphyrkina *et al.* 2001).

Similar to other predators, the persistence of leopard populations is highly dependent on availability of sufficient prey, both preferred prey species and prey of suitable size (Karanth & Sunquist 1995, Hayward *et al.* 2007, Carbone *et al.* 2011). Despite the leopard's broad prey spectrum, reduction of prey availability in their habitats is a key conservation concern in southern Africa (Hayward *et al.* 2007, Henschel *et al.* 2008).

Until 2008, leopard were classified as "Least Concern" on Red List of the International Union for Conservation of Nature (IUCN) due to their wide geographic range (Henschel *et al.* 2008). However, according to Henschel *et al.* (2008), because of population declines in some areas of their range in Africa and tropical Asia, the status of the leopard was reassessed and re-classified as "Near

Threatened" by the IUCN in 2008. The leopard is currently extirpated in at least 36% of their historical range in Africa (Ray *et al.* 2005). In southern Africa, the most dramatic range loss of leopard occurred in South Africa (Henschel *et al.* 2008). These researchers warn that leopard may soon qualify for the Red List status of "Vulnerable" if the current trends continue (Henschel *et al.* 2008).

Currently, there are nine subspecies of *P. pardus* based on genetic analysis (Uphyrkina *et al.* 2001). Of these subspecies, three (Amur leopard *P. pardus orientalis*, Arabian leopard *P. pardus nimr* and Javan leopard *P. pardus melas*) are classified as "Critical" and are facing extinction, and two (Sri Lankan leopard *P. pardus kotiya* and Persian leopard *P. pardus saxicolor*) are listed as "Endangered". Currently leopard inhabit 31 countries in sub-Saharan range states (Ray *et al.* 2005). Mozambique is home to *P. pardus pardus* Linnaeus, 1758 (Henschel *et al.* 2008) and it is one of the range states with the highest leopard abundance (Nowell & Jackson 1996). However, little knowledge of leopard conservation status or population density estimates are available in the country (Purchase & Mateke 2008). Their elusive, solitary and largely nocturnal habits make it difficult to obtain empirical data on leopard populations (Hunter *et al.* 2003).

Leopard are also listed on the Appendix 2.1 of Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in order to address concern of illegal trade in leopard skins (CITES resolution 10:14). A special agreement was made to allow sport hunting of leopard and movement of sport hunting skin trophies. Countries can benefit from leopard hunting to enhance the species' survival (CITES resolution 10:14) and currently 12 countries allow sport hunting of leopard (Balme *et al.* 2010). In 2006, CITES approved the request from Mozambique to double the leopard hunting quota to 120 animals. Since leopard have an important role in the ecosystem (Henschel 2008), and are a key trophy in the sport hunting industry (Balme *et al.* 2010), the relationship between their utilization as trophy and improved species conservation needs to be assessed (Purchase & Mateke 2008).

1.4.2 *Rationale for this study*

The protected area where this study was conducted, Niassa National Reserve (NNR), at 42,000 km^2 represents 5% of the land surface of Mozambique, and comprises 28% and 48% of the country's hunting areas and national parks and reserves, respectively (Magane *et al.* 2009, Soto 2009). The vegetation of NNR is composed of miombo, specifically dry miombo woodland that extends from southern Malawi to Zimbabwe (White 1983). The vegetation is characterized by woody species such as *Brachystegia* species, *Julbernardia globlifora* or *Diplorrynchus condylocarpon* (Ganzini *et al.* 2010).

NNR is an important area for large carnivore conservation (IUCN 2006). It is located in the poorest region of Mozambique (Van den Boom 2011) and is inhabited by approximately 35,000 people who dependent primarily on subsistence agriculture and extractive use of resources for their subsistence (Cunliffe *et al.* 2009). Because of human population growth and associated habitat conversion, there are concerns for conservation of leopard and other large carnivores in NNR (SGDRN 2007).

Consumptive tourism is part of the strategy to generate revenues for conservation management of NNR, and to contribute to the livelihoods of local communities (SGDRN 2007). Hunting in this reserve is based on the principle that wildlife resources can be exploited through licenses issued with clearly regulated terms and without endangering the species conservation, and which are in conformity with the reserve management plan (DNFBB 1999, SGDRN 2007). Leopard is one of the most important trophies among the 26 species available on quota in NNR (SGDRN 2008). More leopard are managed as hunting trophies in NNR than in any other hunting destination in Mozambique (Magane *et al.* 2009). However, there are management concerns related to the provision of equitable and tangible hunting revenues to the local communities (SGDRN 2007). Furthermore, the reserve management acknowledges the need for population density estimates to improve the management of leopard in Niassa reserve (SGDRN 2007).

The case of leopard sport hunting in NNR highlights challenges faced by managers and conservationists trying to protect and conserve carnivores with high value at a global scale but that offer limited revenue at local level in Mozambique and elsewhere. It illustrates the difficulties in distributing sport hunting revenues to encourage people to tolerate the economic losses caused by leopard, and demonstrates the need for better understanding on how humans affect leopard populations through sport hunting or poaching. The main objectives of my thesis were to provide an understanding of leopard hunting as an incentive for local people living inside NNR to coexist and increase their tolerance to leopard, and to suggest possible alternative strategies to increase economic benefits to the communities. I also assessed the conservation status of the leopard population and the pressure they face from both legal and illegal hunting in the reserve. In addition, I provide baseline information on leopard densities and conflict for sustainable management of leopard in NNR.

1.4.5. *Overview of the thesis*

Chapters 2 and 3 have been written as independent papers for publication in peer-reviewed journals. The broader discussion of my findings and a demonstration of their wider application to carnivore conservation and management are presented in Chapter 4. Lastly, I present a full list of references cited in this thesis. Below I present a brief description of each chapter.

In Chapter 2, I analyzed the economic benefits from leopard sport hunting and local trade of their skins in relation to the costs from goat depredation in four villages. I interviewed all NNR hunting concessionaires (n = 8) and a sample of villagers (n = 158) on the importance and economic value of leopard. I also investigated the cost of depredation on livestock by leopard, and the illegal trade of skins in the village. These analyses are useful to understand the distribution of benefits from leopard hunting at different scales. I used this information to propose pragmatic measures to increase the economic sport hunting benefits allocated to communities.

In Chapter 3, I assessed leopard density and relative abundance of their prey in miombo and riparian habitats. Using camera trapping, I calculated and compared leopard population densities in two habitats (four study sites) and the relative abundance of their prey. In addition, I analyzed the turnover of leopard over two and three years and assessed the sustainability of legal and illegal off-take of leopard in NNR. Finally I discussed possible anthropogenic effects on leopard densities, and the management implications of my results.

CHAPTER 2

Costs and benefits of leopard to the sport hunting industry and local communities in Niassa National Reserve, Mozambique

2.1. Summary

Sport hunting is increasingly proposed as a tool to support the conservation of large carnivores. However, it is challenging to provide tangible economic benefits from the sustainable use of large carnivores that can be used as an incentive for local people to conserve the predators living alongside them. The relative importance of leopard *Panthera pardus* to the sport hunting industry and to the local communities in Niassa National Reserve (NNR), Mozambique, was investigated by comparing the economic gains from leopard sport hunting and poaching with the economic losses from predation of goats. Leopard are the mainstay of the hunting industry, especially in the western part of NNR with its relatively high human population density and low abundance of other key trophies. They are highly valued by tourist hunters, with a single trophy worth approximately US$24,000 but skins are illegally traded locally for small amounts ($83). Leopard depredated 11 goats over two years in two villages, causing monetary losses of $440 to six households. Although goat depredation contributed to peoples' negative attitudes towards leopard, poaching appears to be driven more by the economic value of leopard skins than the damage caused by leopard. We propose Payments to Encourage Coexistence (PEC) fund from the tourism related to large carnivores to improve the flow-on benefits to people affected by leopard and to effectively compensate for human injuries and deaths and livestock losses, caused by large carnivores. The long-term survival of leopard and other large carnivores in NNR and elsewhere will depend on the flow of tangible conservation benefits to communities to build tolerance and positive actions towards predator conservation.

2.2. Introduction

Wildlife conservation is a costly activity and often competes with other societal priorities (Joseph *et al.* 2008). Therefore for conservation to succeed, it should include incentive driven strategies that support conservation in areas where humans live alongside dangerous animals (Hutton & Leader-

Williams 2003). These strategies must have the potential to not only achieve biodiversity conservation, but also achieve poverty alleviation goals (Abensperg-Traun 2009). The potential for economics to inform such conservation decisions through a cost-benefit approach is clear (Naidoo *et al.* 2006).

Despite consensus on the need to conserve African wildlife for future generations, no agreement exists on the most appropriate strategy to achieve this goal (Baker 1997). The conservation community, general public, and governments at national and international levels are divided between consumptive and non-consumptive use strategies (Novelli *et al.* 2006). Some countries, such as Kenya, have adopted an exclusively non-consumptive use policy while others, like South Africa and Namibia, have been successful in combining both strategies (Lindsey *et al.* 2006). With either strategy, the challenge is to generate revenues sufficient to maintain and protect wildlife and compete with other land uses.

One situation where an economic-based analysis is particularly useful is in determining the value of sport hunting to stakeholders (Leader-Williams 2009). Proponents of sport hunting argue that well managed sport hunting can have positive economic and ecological effects on community livelihoods and wildlife conservation (Loveridge *et al.* 2006, Lindsey *et al.* 2007a). Sport hunting can be a conservation tool when local communities tolerate large populations of potentially dangerous animals in exchange for revenues from the harvest of some proportion of the population (Leader-Williams 2009). Sport hunting can also be a conservation tool when a considerable proportion of the revenues earned are invested in species conservation (Balme *et al.* 2010). However, despite the stated benefits, an assessment of the conservation role of sport hunting in many parts of Africa is hampered by lack of data on the economic impact of the activity (Lindsey *et al.* 2006) and an objective assessment of whether sport hunting is improving in situ community livelihoods and wildlife conservation.

In Mozambique, many protected areas are under-funded and few are able to meet their conservation goals (MITUR 2003). In Niassa National Reserve (NNR) conservation management activities are significantly funded through sport hunting revenues (SGDRN 2007). NNR is one of the major hunting destinations in Mozambique, but also supports a growing human population of 35,000 people (INE 2008a, 2008b). Legally, approximately 20% of the revenues from trophy fees must accrue to villages within NNR, ostensibly as an incentive for tolerating wildlife in the reserve

(DNFFB 1999). Whether this revenue from trophy licenses adequately compensates for the costs of living with wildlife has yet to be economically determined.

In this paper we assess the economic benefits to stakeholders from sport hunting and poaching of leopard *Panthera pardus*, and the economic costs to communities that live with leopard in NNR. We focus on leopard because it is economically important for both photographic and sport hunting operations (Lindsey *et al.* 2006, 2007b), and is also the main cause of conflict with livestock, primarily goats in NNR (Begg & Begg 2007). In terms of conservation, leopard are listed on Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) because they are declining in many range states owing to habitat loss and fragmentation, and hunting for trade and predator control (Packer *et al.* 2009). The hunting of other large carnivores, such as lion *P. leo* and spotted hyaena *Crocuta crocuta*, but not African wild dog *Lycaon pictus*, are also permitted in NNR (DNFFB 1999).

Detailed in-country studies are needed where hunting occurs to allow for improved assessment of the conservation role of hunting as well as to identify problems and find appropriate, site-specific solutions (Lindsey *et al.* 2007a). However, assessing the contribution of sport hunting to development in Mozambique is constrained by the paucity of data (Magane *et al.* 2009). Our study aimed to partially fill this gap, and to contribute to the current debate on sport hunting. We specifically addressed: (1) the economic importance of leopard to the NNR sport hunting industry, (2) the economic benefits accruing from leopard sport hunting and poaching, (3) the costs of leopard depredation of livestock to local communities, and (4) the attitudes of communities towards leopard. Finally, we proposed ways in which the benefits from leopard sport hunting can be used to improve community livelihoods, and provide incentives for sustainable species conservation.

2.3. Study area

The study was conducted in NNR, Mozambique's largest protected area (42,000 Km^2; -12.1670S, 37.5450E), located on the border with Tanzania along the Rovuma River. As one of the largest protected Miombo forest ecosystems in the world, NNR supports a wide range of large mammal species which, in many cases, comprise the largest populations within Mozambique (Agreco 2008, Craig 2009). NNR was originally proclaimed as a Game Reserve in 1954 and is managed by Sociedade para a Gestão e Desenvolvimento da Reserva do Niassa (SGDRN), a partnership between government and a private company, Investimentos Niassa Ltd. NNR has designated concessions for hunting (9 units), photographic tourism (6 units) and resource conservation (2 units). One of SGDRN's goals is to provide a quality ecotourism and hunting experience to generate a local economy to finance management activities (SGDRN 2007).

Census data suggest relatively high rates of human population growth and expansion of the 40 settlements currently within NNR over the last ten years (INE 2008a, 2008b). The resident communities are characterized by a high degree of poverty, poor food security, poor access to infrastructure and services, and high levels of conflict with wildlife, particularly African elephant *Loxodonta africana* and large carnivores (Cunliffe *et al.* 2009). Subsistence farming, characterized by shifting cultivation, is the principal livelihood activity (Cunliffe *et al.* 2009). The utilization of resources, including consumptive use of wildlife, tends to act in direct opposition to the stated conservation goals of the reserve (Cunliffe *et al.* 2009). Cattle are absent in NNR due to tsetse fly *Glossina* spp., but goats and chickens, and domestic dogs and cats are present in the villages. The overall estimated livelihood contribution from fishing, livestock, trading, and agriculture ranges from US$5 to US$1,532 per household per annum (Cunliffe *et al.* 2009).

2.4. Methods

We identified the key stakeholders in the NNR hunting industry and determined components of the value chain for the cost of hunting (as per Booth 2009, Fig. 2.1). We calculated revenues for stakeholders based on the direct expenditure by clients on hunting packages, travel and accommodation, and trophy export and taxidermy (Appendix 1).

The multiplier effect of client expenditure from sport hunting includes the payments by hunting operators to SGDRN for concession fees, salaries and other benefits allocated to local communities, professional hunters' (PH) guide licenses to the Ministry of Tourism (MITUR), firearm and ammunition licenses for the PH and work visas to the Ministry of Interior (MINT). The government taxes from salaries of concessionaire workers and revenues from the local petrol station were also considered part of the multiplier effect of client expenditure.

The Forestry and Wildlife law (DNFFB 1999) stipulates that 20% of revenue generated from all safari and other tourism activities in NNR must be transferred to local communities. We therefore considered the 20% from revenues of the MITUR trophy fee and SGDRN royalty and concession fees accrued to communities as part of the multiplier effect of clients' expenditure. To compare the sport hunting revenues with benefits from poaching and costs from conflict accrued to households, we divided the community hunting benefits by the 40 villages of NNR multiplied by 420 households estimated for Mbamba village (Niassa Carnivore Project (NCP) unpublished data).

Interviews regarding the value of leopard were conducted with respondents from the NNR hunting concessions and villagers in 2010 and 2011. To determine the economic benefits from leopard hunting, we sent e-mail questionnaires ((in English, Appendix 2) to all NNR hunting operators (n = 8) in February 2011. These questionnaires were written to determine the economic value of leopard for each operator, and to assess the importance of leopard compared with buffalo *Syncerus caffer*, lion and elephant for the industry. Our sample included all current hunting operators in NNR, and was therefore a complete representation of the economic value of leopard and its importance to the hunting industry in NNR. Due to lack of information, expenses such as food for hunting clients, salaries for senior staff, equipment acquisition, maintenance, marketing, and office expenses were not included in our analysis of costs covered by economic benefits from leopard hunting.

Between September and December 2010, we conducted questionnaire surveys in villages inside NNR to assess the perceived value of leopard and costs from leopard predation on goats. Although there are economic benefits from leopard poaching, it is an unlawful activity in Mozambique (DNFFB 1999) that people are generally averse to address. To provide an index of the numbers and prices for leopard skins illegally traded in NNR against which to assess the accuracy of questionnaire data from villagers, we first interviewed a former poacher.. This individual was a local resident in Mbamba village in NNR and was employed by a conservation project to assess lion and leopard mortalities in snares. He provided us with an estimate of the number of leopard skins traded in Mbamba village in 2010 and the market price for these skins.

Secondly, we used structured questionnaires (conducted in the national language, Portuguese, Appendix 3), Prior to the questionnaire survey of villagers (conducted in the national language, Portuguese, Appendix S3), we held one meeting in every village to present the objectives of the interviews and explain the importance of participation to the community and local leaders. It was made clear that the objective was not to assess illegal use but to understand the benefits and costs of leopard to the community. We conducted the survey in collaboration with the resident community agent of the Management Orientated Monitoring System (MOMS, a community based system to collect data, including on human wildlife conflict). These agents are familiar with data collection as part of their MOMS duties and were able to translate the questionnaire into the local languages, Cyao, Cmakua or Swahili, where necessary. During the village questionnaire survey, we asked people to estimate the numbers and values of leopard killed illegally in four villages inside NNR during that year (Mbamba: n = 24 interviews; Mussoma: n = 52, Macalange: n = 32; Lissongole-Cuchiranga: n = 50). These four villages were chosen as there were existing data on human-carnivore conflict collected by MOMS agents from 2007 onwards, and there was evidence of illegal hunting of carnivores. Sample sizes were uneven as the villages differed in size and different numbers of people were available to interview across the villages.

We defined the costs of living with leopard as the economic losses from leopard depredation of goats in NNR. We used our village questionnaire to estimate the numbers of goats owned by households, the relative proportion lost to leopard predation, and to assess livestock husbandry techniques of the local communities. We used unpublished data from the NCP and MOMS on human-leopard conflict in NNR from 2009 to 2010 to verify the local reports of livestock killed by leopard in the four villages. Cost of goat depredation was calculated based on their market value in the villages (A. Jorge personal observation). We compared the differences in livestock husbandry techniques used by villagers with their attitudes towards leopard. Our analysis does not include the economic losses due to injuries from leopard on people or from loss of other domestic animals such as chickens, which was minimal.

2.5. Results

In 2010, a total of 70 hunting clients (mostly from USA, 55% and France, 13%), and 26 observers, spent a total of US$3,286,688 over 1,070 days while sport hunting in nine concessions in NNR (Table 2.1). The expenditure from clients was predominantly for hunting packages (47%), accommodation and travel (23%), trophy shipping and mounting (19%), and for government and SGDRN hunting fees (11%; Fig. 2.1). Earnings by SGDRN ($481,934) from trophy and concession fees were invested in anti-poaching and management activities in NNR, covering 30% of the annual operational costs of the reserve. Through the multiplier effect, communities received revenue from sport hunting through employment ($130,763; 41%), 20% of the trophy and concession fees ($122,568; 38%), and direct purchasing by concessionaires of local materials for construction e.g. thatching grass, bamboo, etc. ($67,347; 21%; Table 2.1).

Of the 19 safaris that included leopard hunting (27%) in 2010, leopard were the main trophy in 16 safaris, while in three safaris leopard were hunted with other key species such as lion or elephant. On average, five ungulates were shot on leopard safaris as secondary trophies, bait or meat rations for concession staff and villagers. In total 16 hunters (from USA, 69%; Italy, 13%; Germany, and Mexico and Portugal, 6% each) expended $927,353 over 225 hunting days on leopard hunting with an additional $25,940 in revenues through the multiplier effect (Fig. 2.2). The mean length of each safari was 14 days with an average client expenditure of $2,587 per day per client on government taxes, accommodation, safari expenses, trophy handling and packing, and air charter. The overall direct expenditure on leopard safari package ($431,888) was 28% of expenditure on all hunting packages.

Most of the revenues from leopard hunting were retained by hunting concessionaires (47%), taxidermists (24%) and travel agencies (13%; Fig. 2.2). According to the hunting operators (n = 7), the mean (SE) economic value for a leopard trophy was $23,878 (1,375), but the amount varied (range: $20,000 – $30,000) depending on the operator. Nevertheless, all concessionaires derived a larger proportion of revenue from the daily rate ($17,200 – $25,000) than from the trophy fee ($4,100 – $6,000). Although the average client expenditure for leopard hunting ($25,997) was less than that for elephant ($47,067) or lion ($65,255), because more leopard safaris (n = 16) were conducted compared to lion (n = 4) or elephant (n = 2), it led to higher gross revenues for concessionaires (Fig. 2.3a). In the western part of NNR with relatively high human population density and relatively low wildlife abundance, leopard safaris contributed 33% of the total operators' (n = 4) revenue ($712,597). In the eastern portion of NNR, which is less inhabited and has a relatively high-density of wildlife, leopard safaris were the second major contributor (23%) to the operators' (n = 3) gross revenues ($806,273) following buffalo.

Leopard were generally perceived as the second most important species, following elephant, for the hunting business of four concessionaires (Fig. 2.3b). Only one operator, with no previous experience in NNR, ranked leopard as the least important species. Based on the analysis of annual reports, the total revenue from leopard safaris (Table 2.1) was equivalent to 90% of the operators' annual costs of salaries ($435,326) for local workers, government taxes, fuel, and concession fees.

There was a large difference between the estimated revenue generated from leopard poaching and leopard sport hunting in NNR (Table 2.1). In 2010, at least two leopard skins were reported by a former poacher to be traded from Mbamba village, and each sold for ~$83 to outsider trader(s). The local communities were understandably reluctant to discuss leopard poaching, and predictably many respondents (82%, n = 130 questionnaires) provided no estimation of the numbers of leopard killed in their villages. Of those that did respond (n = 28), two of the four surveyed villages provided an estimate of up to four leopard poached during the past year. This estimate is supported by the observation that three radio collared female leopard were snared by people from Mbamba in the same year (C. Begg, personal observation). Almost 80% of respondents provided estimates for the economic value of a leopard skin, with 33% estimating the value as less than $100.
When asked about leopard depredation on livestock, only six livestock owners (23% of 26 owners) reported losses of goats to leopard between 2009 and 2010. For these six households the monetary cost of depredation totaled $440 in two years, with $73 lost by each household. Despite the low occurrence of leopard attacks on goats, the loss was high for individual households when it occurred. In addition leopard depredation affected the attitudes of people towards the felid. More people from villages where leopard depredated goats in the past two years reported negative attitudes towards leopard compared to areas with no recent records of leopard attacks (Table 2.2). When asked about possible measures to prevent leopard depredation on goats, villagers either stated that they did not know how to prevent leopard attacks (27%), or stated that goats should be kept in corrals at night (58%), leopard should be removed from NNR (9%), and domestic dogs should be used to protect goats (7%). People with positive attitudes toward leopard (61%, n = 56) were more

likely to propose corralling to protect the animals than those with negative attitudes ($\chi^2 = 42.41$, df = 2, $p < 0.01$). All people who proposed drastic measures such as removal of leopard from the reserve to prevent attacks on goats were indifferent (n = 1) or disliked leopard (n = 13). In practice, 85% of goat owners (n = 22) reported the use of corrals, with the remaining using unfenced yards as their livestock husbandry technique. Of the goats killed by leopard, seven were inside corrals and four were in unfenced yards.

In response to the question of the importance of leopard, villagers valued leopard mainly for bringing hunting and photographic tourists (58%, n = 91), for preying on crop-raiding wild animals (15%, n = 24), for its skin (10%, n = 16), and for the usefulness of its body parts for medicinal purposes (3%, n = 4). Some villagers claimed to not know the importance of leopard (13%, n = 21) or considered the animal unimportant (1%, n = 2).

2.6. Discussion

This study represents the first assessment of the contribution of sport hunting revenues to conservation at local and national levels in Mozambique. Leopard were the most important contributor to the sport hunting industry, and the overall revenues from legal hunting of leopard far exceeded revenues from poaching. However, the potential economic benefits accruing to individual households from leopard hunting (< $1) and sport hunting in general ($19) are much less than losses from goat depredation and benefits from poaching.

Sport hunting generates large sums in southern Africa (Lindsey *et al.* 2007a), and is an important source of revenue for Mozambique (Magane *et al.* 2009). Similar to the findings of Balme *et al.* (2010) for South Africa, the leopard is one of the most important species in the Niassa National Reserve (NNR) sport hunting industry. The average daily expenditure per leopard hunter ($2,587), who stayed for an average 14 days, was six times higher than that spent by photographic tourism clients in NNR ($440) accommodated for an average of 2.5 days (SGDRN unpublished data). This daily expenditure was also higher (59 times) than the conventional daily expenditure ($44) of tourists in Mozambique (INE 2008c). The daily expenditure of clients hunting leopard in NNR was double the daily expenditure ($1,270) of other hunters as estimated from the assessment of sport hunting in the Namibian economy (Humavindu & Barnes 2003). In fact, leopard trophy fees in Mozambique are the third highest in southern Africa, following Botswana and Tanzania (Booth 2009), and NNR has the highest fees in the country (A. Jorge unpublished data). Yet, NNR operators rely more on the daily rates from leopard hunting to earn their income, rather than on trophy fees. By contrast, 60% of revenues from the Wildlife Division in Tanzania are earned from trophy fees (Baldus & Cauldwell 2004).

The low human-leopard conflict and relatively few livestock depredated in NNR suggest that leopard are generally poached because of the economic value of their skins rather than in retaliation for livestock depredation. A single leopard skin provides an amount equivalent to one month's salary ($83, SGDRN unpublished data) in NNR at almost no cost to poachers. Although this is much less than the average revenue generated from the sale of ivory from a single poached elephant ($2,500, SGDRN unpublished data), this is still an important source of income for a local hunter (Kühl *et al.* 2009). Furthermore, poachers can seldom afford to pay the minimum fine for leopard poaching ($453, DNFFB 1999). Increasing opportunity costs through more effective anti-poaching (Messer 2000), and providing alternative sources of income may contribute to effectively reducing poaching (Kühl *et al.* 2009). For example, in NNR, employing local people to pack hunting trophies for trophy shipment companies could potentially provide the same level of earnings as leopard poaching.

Similar to NNR, attacks by large predators to livestock inside corrals were reported elsewhere (Kenya, Ogada *et al.* 2003; India, Namgail *et al.* 2007; Bhutan, Wang & Macdonald 2006; and Nepal, Oli *et al.* 1994), mostly in poorly designed corrals, built to enclose goats rather than exclude predators (Namgail *et al.* 2007). The estimated annual per capita loss from depredation of livestock was lower in NNR (8% of $414 for 2008, United Nations Statistics Division 2011) compared with Nepal (25%, Oli *et al.* 1994), India (18%, Namgail *et al.* 2007) and Bhutan (17%, Wang & Macdonald 2006). Despite the low occurrence of livestock depredation in NNR, people from affected villages were negative toward leopard. Our attitudinal results contradicted findings from India, which instead report higher positive attitudes towards Snow leopard (*P. uncia*) by villagers who lost more livestock (Bagchi & Mishra 2006). This difference is most likely a function of how little those communities depend on livestock for direct income (Bagchi & Mishra 2006). In NNR, goats are the most valuable livestock and few villagers can afford to buy them (Cunliffe *et al.* 2009). Thus, a successful resolution of the issue of livestock predation is important for the subsistence of communities, and for carnivore conservation (Ogada *et al.* 2003, Treves & Karanth 2003, Begg & Begg 2011).

The depredation of livestock by carnivores inside corrals can be effectively mitigated in NNR and elsewhere by improving livestock husbandry in the villages (Jackson & Wangchuk 2004, Begg & Begg 2011) and by providing incentives to the communities (Namgail *et al.* 2007). Since hunting operators can purchase goats as bait animals from villages, improved corralling methods could increase the annual income of local households ($40 per goat) and would be most effective if directed towards villagers with no formal employment. By using community owned goats to supplement ungulate baits on leopard safaris, hunting operators could also use the substituted wild ungulates on plain's-game safaris, thereby increasing their revenues. This is especially relevant as market resistance to the carnivore hunting packages grows (Booth 2009) and more clients can afford to pay for plain's-game safari packages.

The high value of leopard for trophy hunters compared with the low value for communities poses a serious challenge for leopard conservation (Dickman *et al.* 2011), particularly in light of continued human population growth (INE 2008a, 2008b) and related threats to populations of large carnivores in NNR (Begg & Begg 2010). It is therefore critical to provide incentive driven conservation to people sharing land with dangerous animals such as leopard (Hutton & Leader-Williams 2003). Although the proportions of hunting revenues accrued to NNR communities (20%) are higher than those allocated to villages in Zambia (12%; Lindsey *et al.* 2007a) and Cameroon (10%; Yasuda 2011), there is still a need to increase the revenues and improve the benefit sharing among the communities within the reserve. To improve coexistence with dangerous large carnivores, communities should get tangible and direct benefits from sport hunting to compensate for the costs imposed by the presence of predators in their areas (Packer *et al.* 2009). Furthermore, the political instability in Zimbabwe (Booth 2005), limited availability of wild areas for sport hunting large carnivores in South Africa (Lindsey *et al.* 2007a), and the moratorium on lion hunting in Botswana (Packer *et al.* 2009), may lead to an increased demand for carnivore trophies from NNR and the rest of Mozambique in the near future. Therefore, it is critical to develop effective economic incentive structures, especially for local communities (Albensperg-Traun 2009), if both the large carnivore populations and sport hunting are to persist in Mozambique. For instance, carnivore populations in communal lands in Namibia are increasing, mainly because people receive substantial revenues from their sport hunting, and consequently value them (Frank 2010). By contrast, expanding human populations and the associated problems in sport hunting management has lead to a decline of lion and leopard populations in some areas in Tanzania (Packer *et al.* 2011).

Recently, Dickman *et al.* (2011) presented a framework, the Payments to Encourage Coexistence (PEC), which has potential to meet both social and ecological objectives for carnivore conservation. We believe that stakeholders of the hunting industry in NNR can incorporate a similar scheme to generate funds, and finance the strategies needed to improve local livelihoods and foster large carnivore conservation (Fig. 4). By giving user rights to NNR communities, they could operate leopard safaris in partnership with local operators and retain most of the safari revenue for the fund (V. Booth personal communication). A similar approach was proposed for farms with losses of livestock to leopard in Namibia (Stein *et al.* 2010), and this strategy has been key to the conservation of jaguar (*P. onca*) in human dominated landscapes in Mexico (Rosas-Rosas & Valdez 2010). Furthermore, through agreements with NNR operators, reputable taxidermists and travel agencies in the US and European markets could be granted exclusive rights to provide services to NNR clients, in return for contributing a certain percentage of their profits to the PEC. Hunting clients are prepared to support conservation initiatives (Lindsey *et al.* 2006), therefore an additional fee ($100 – $200 per client) could be charged to finance the PEC. Photographic tourism clients usually outnumber hunting clients (Lindsey *et al.* 2007b) and can also be incorporated in the PEC. Following Salafsky & Wollenberg (2000), economic incentives proposed for local communities in our scheme are directly linked with large carnivore presence to encourage people to conserve them.

2.7. Conclusion

Leopard are one of the most important trophy species in the hunting industry and contribute substantially, both directly and in multiplier terms, to development in NNR and Mozambique. The community livelihood concerns in NNR can be better addressed by increasing the size of sport hunting benefits and improving the distribution of these revenues to communities, and by better resolution of human leopard conflict through improved livestock husbandry. Although leopard are killed primarily by sport hunters in NNR, additional harvest pressure is exerted because of illegal markets for their pelts outside NNR. Poaching is not only an ecological concern but also an economic loss given the importance of leopard to the hunting industry, and the importance of the revenue derived from hunting for conservation management. Based on the current scenario, it is unrealistic to expect villagers to stop hunting leopard for their skins when the need for income is immediate. The shift from leopard poaching, with short-term profit, to sustainable sport hunting with tangible benefits for local people, is a clear challenge for the conservation of these felids in NNR. These goals may be better achieved with coordinated efforts between the stakeholders in the sport hunting industry, managers, conservationists, and civil society. To ensure that leopard related tourism activities provide tangible benefits that off-set existing costs and foster conservation in NNR, mechanisms such as PEC could be incorporated into the management paradigm.

Table 2.1. The revenues generated (US $) from hunting clients and their observers from all hunting safaris and leopard safaris, and from leopard poaching in one year (2010). The multiplier effect refers to the distribution of clients' moneys to government institutions, SGDRN, local communities and petrol stations in 2010.

Stakeholder	**All hunting safaris (n = 70)[a]**		**Leopard safaris (n = 16)**		**Leopard poaching (n = 8)[i]**
	Direct expenditure	**Multiplier effect**	**Direct expenditure**	**Multiplier effect**	
MITUR[b]: trophy fee	130,907	0	21,168	0	0
MITUR: PH and client licenses	3,838	0	515	0	0
MITUR: export permits	483	0	133	0	0
MINT[c]: firearm licenses	15,088	3,500	3,142	2,000	0
MINT: work and tourist visas	28,580	16,935	2,295	8,085	0
MINAG[d]: export permits	3,380	0	2,523	0	0
MINF[e]: salary taxes	0	8,095	0	0	0
Government	**182,276**	**28,530**	**29,776**	**10,085**	0
Trophy fee	575,257	0	184,308	0	0
Daily rate	965,622	0	247,580	0	0
Operators	**1,540,879**	**0**	**431,888**	**0**	**0**
Royalty fee	186,727	0	32,081	0	0
Concession fee	0	295,207	0	0	0
Trophy fee	0	0	0	0	0
SGDRN[f]	**186,727**	**295,207**	**32,081**	**0**	**0**
Trophy fee	0	63,527	0	10,650	0
Concession fee	0	59,041	0	0	0
Employment[g]	0	130,763	0	0	0
Meat[h]	0	51,347	0	0	0
Construction materials	0	16,000	0	0	0
Local hunter(s)	0	0	0	0	670
Communities	**0**	**320,678**	**0**	**10,650**	**670**
Air charter companies	205,100	0	61,700	0	0
Hotels	37,800	0	9,100	0	0

Travel agencies	515,000	0	121,500	0	0
Private custom agents	2,501	0	493	0	0
Trophy shipment companies	41,175	0	14,535	0	0
Taxidermists	575,230	0	226,280	0	0
Petrol stations	0	252,112	0	9,975	0
Private sector	**1,376,806**	**252,112**	**433,608**	**9,975**	**0**
TOTAL	**3,268,688**	**896,527**	**927,353**	**30,710**	**670**

[a]amount generated from trophy hunting of 360 animals of 26 species on quota in 2010: buffalo *Syncerus caffer*, common duiker *Sylvicapra grimmia*, suni *Neotragus moschatus*, red duiker *Cephalophus natalensis*, blue duiker *C. monticola*, oribi *Ourebia ourebi*, steenbuck *Raphicerus campestris*, reedbuck *Redunca arundinum*, wildebeest *Connochaetes taurinus*, crocodile *Crocodylus niloticus*, kudu *Tragelaphus strepsiceros*, eland *Taurotragus oryx*, elephant *Loxodonta africana*, warthog *Phacochoerus africanus*, hartbeest *Alcelaphus lichtensteinii*, helmeted guineafowl *Numida meleagris*, spotted hyena *Crocuta crocuta*, hippo *Hippopotamus amphibious*, bushbuck *T. scriptus*, impala *Aepyceros melampus*, waterbuck *Kobus ellipsiprymnus*, lion *Panthera leo*, leopard *P. pardus*, yellow baboon *Papio cynocephalus*, sable *Hippotragus niger*, bushpig *Potamochoerus larvatus* and zebra *Equus quagga*.

[b]Ministry of Tourism.

[c]Ministry of Interior.

[d]Ministry of Agriculture.

[e]Ministry of Finances.

[f]Sociedade para a Gestão e Desenvolvimento da Reserva do Niassa (Society for Management and Development of Niassa Reserve).

[g]salaries paid to approximately 400 temporary and seasonal local workers.

[h]market value of ~ 4500 kg of meat distributed as ration to concession workers and to the villages, and money paid to communities to purchase buffaloes from community quota.

[i]value calculated from our baseline study in Mbamba village.

Table 2.2. The number of goats owned and depredated by leopard and the attitude of people towards leopard in four villages surveyed in 2010 in NNR.

Village	Goats* owned	Goats depredated	Attitude towards leopard		
			Like	Dislike	Indifferent
Lissongole-Cushiranga	1 (50)	1	26%	68%	6%
Macalange	0 (32)	0	75%	25%	0%
Mbamba	0 (24)	0	79%	8%	13%
Mussoma	51 (52)	10	33%	63%	4%
Total	**52**	**11**			

*N = number of people surveyed in brackets

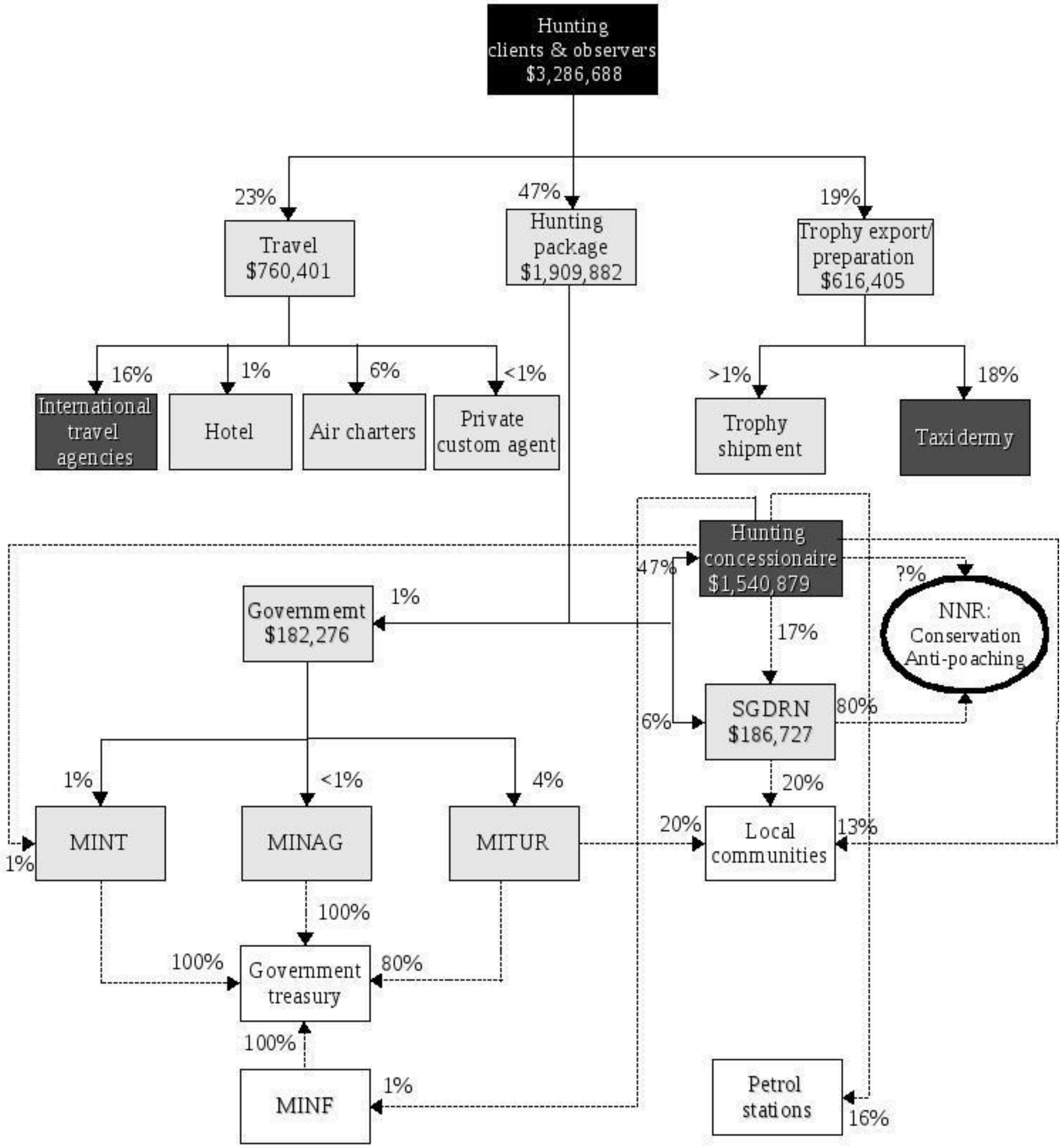

Figure 2.1. Distribution of expenditure from hunting clients and observers ($3,268,688 n = 70) to stakeholders in the NNR hunting industry in 2010. Acronyms: MITUR, Ministry of Tourism; MINT, Ministry of Interior; MINAG, Ministry of Agriculture; MINF, Ministry of Finances; SGDRN, Sociedade para a Gestão e Desenvolvimento da Reserva do Niassa (Society for Management and Development of Niassa Reserve). Solid lines reflect direct expenditure (> 10%, dark grey; and < 10%, light grey) and dotted lines represent multiplier effect of the expenditure.

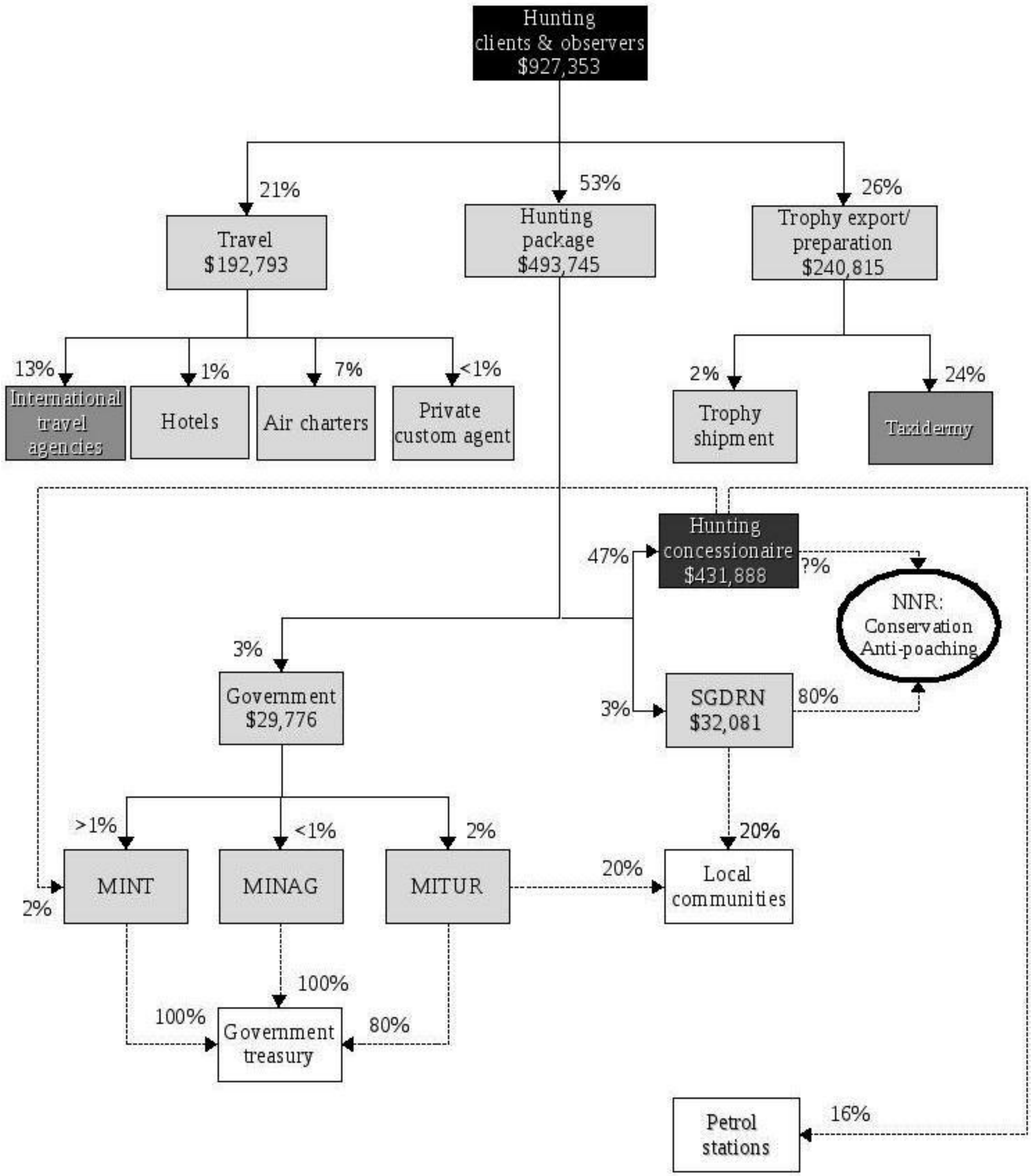

Figure 2.2. Distribution of expenditure from leopard hunting ($927,353, n = 16) in NNR in 2010. Solid lines reflect direct expenditure (> 10%, dark grey; and < 10%, light grey) and dotted lines represent multiplier effect of the expenditure.

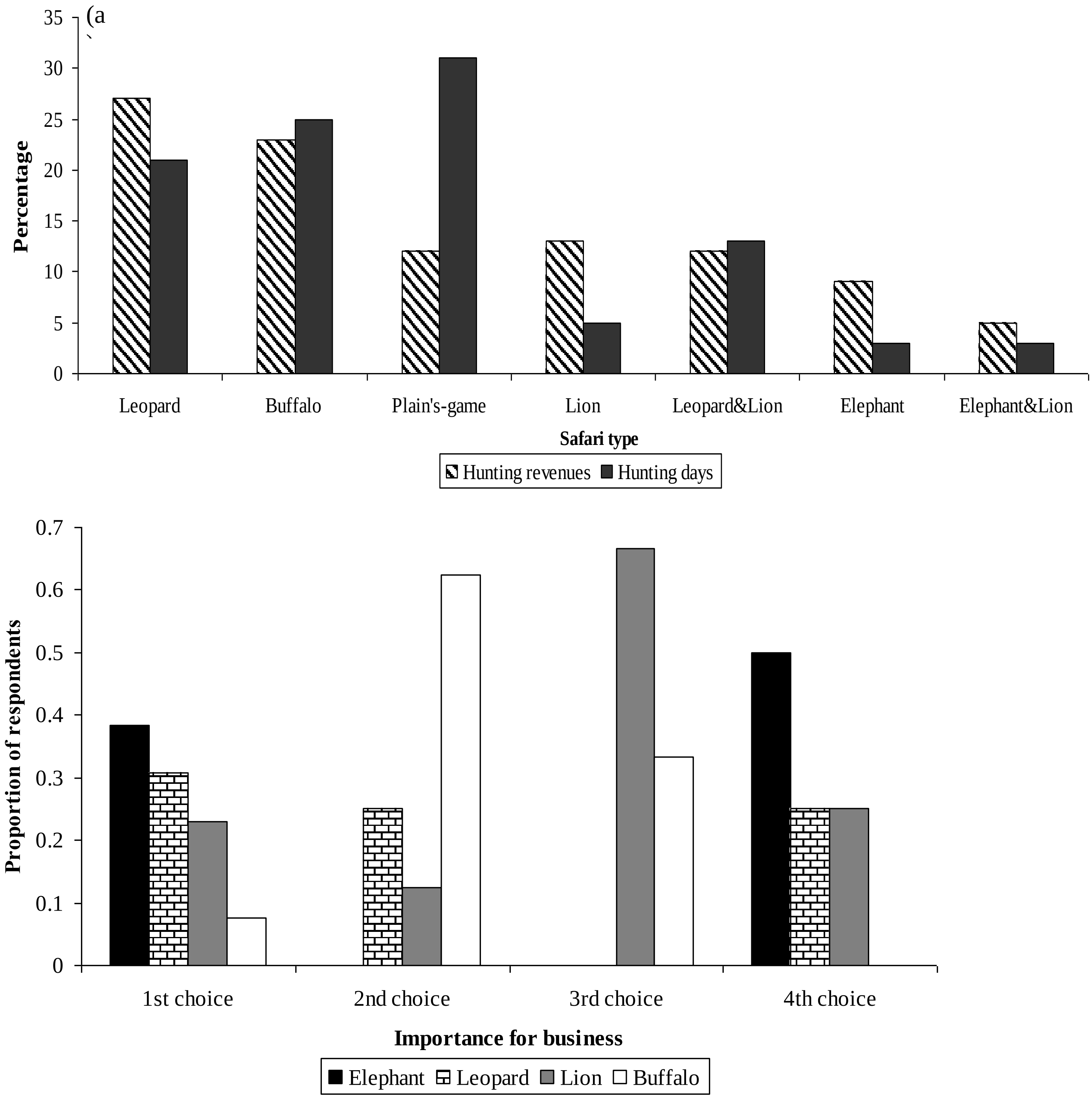

Figure 2.3. Percentage of revenue to the operators from trophy fees and hunting days from safaris during hunts of lion, leopard, buffalo, elephant and ungulates in NNR in 2010; (b) Importance of trophies for hunting business ranked by operators (n = 7) in NNR

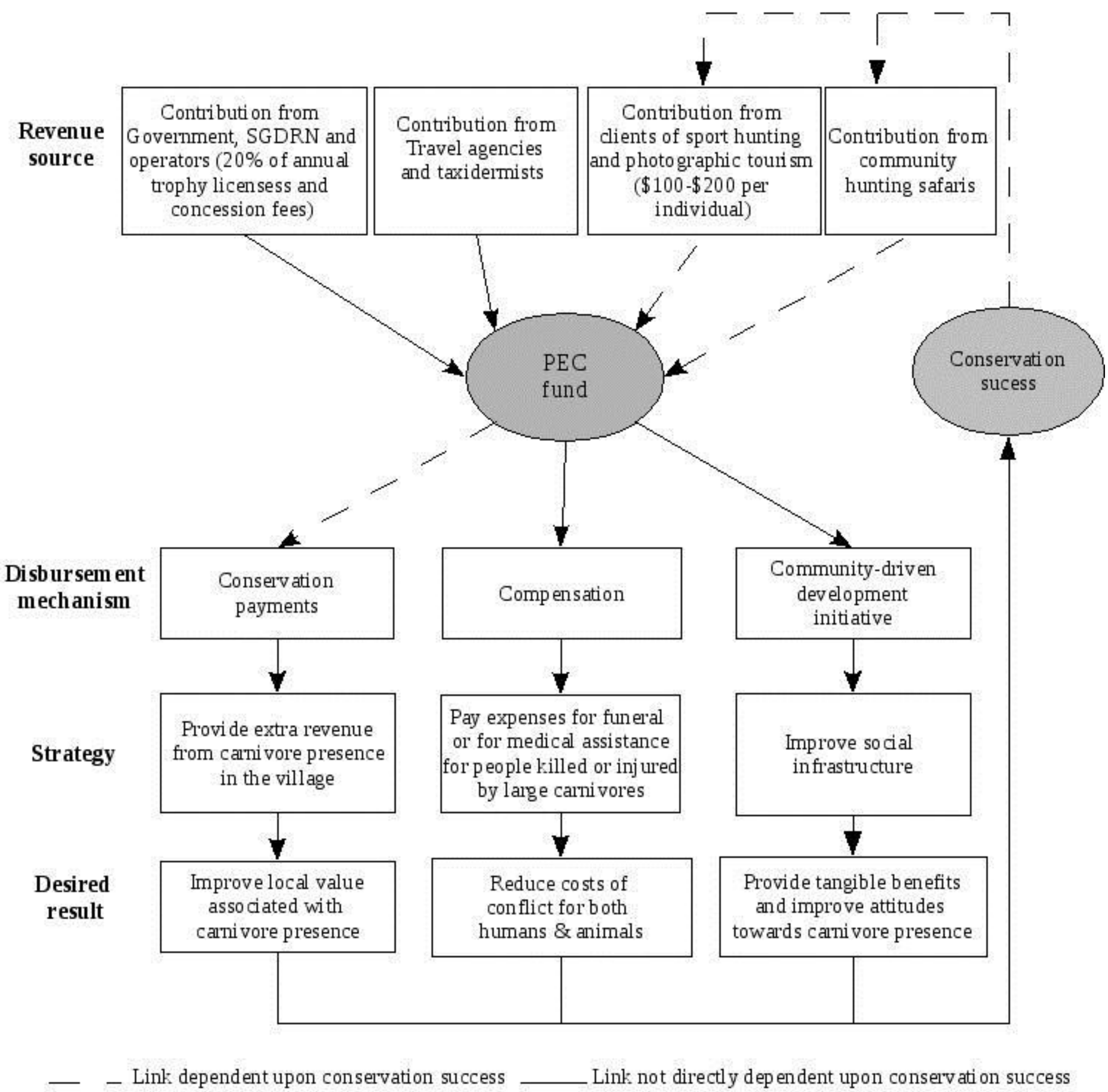

Figure 2.4. PEC scheme to encourage carnivore conservation in NNR. Modified from Dickman *et al.* (2011), with permission.

CHAPTER 3

Leopard densities, hunting quotas and off-take in a miombo ecosystem: Niassa National Reserve, Mozambique

3.1. Summary

To manage consumptive use programs such as sport hunting of leopard *Panthera pardus* and other large carnivores, and ensure sustainability, it is critical to understand the ecology of carnivore populations. Analysis of three years (2008 – 2010) of camera trapping data, combined with seven years of leopard hunting data (2004 – 2010), allowed us to assess the status of leopard populations and the sustainability of leopard hunting in the Niassa National Reserve (NNR) in Mozambique. Compared with miombo woodland areas with 2.18 – 4.31 leopard/100 km^2, riparian study areas had higher relative abundance of potential prey, and supported higher leopard densities (9.92 – 12.65 leopard/100 km^2). Overall, the estimated leopard population densities in NNR (2.18 – 12.65 leopard/100 km^2) were comparable to estimates reported from other study sites in southern and central Africa. Riparian study sites showed a decline in leopard population density over the three year study period, and all leopard photo-captured in 2010 were new individuals, suggesting high turnover rates in this habitat. The mortality from illegal leopard hunting (0.375 leopard/100 km^2) was additive to the mortality from hunting operators (0.068 leopard/100 km^2), and resulted in an unsustainable off-take. Monitoring sport hunting off-takes, and zoning community areas and wildlife habitats, will be critical to minimize the adverse impacts of sport hunting, poaching and continued human population growth, on leopard populations in NNR.

3.2. Introduction

Estimates of population abundance and trends of large terrestrial carnivores are needed to inform conservation actions and ensure their sustainable management (Balme *et al.* 2007, Funston *et al.* 2010). However, due to their elusive nature, and the spatial and temporal scale of carnivore movements, it has been difficult to obtain reliable estimates for these species (Karanth *et al.* 2006). Since the development of robust statistical models to accurately estimate the population density of large felids from capture/recapture models (Karanth and Nichols 1998), camera trapping has been widely applied to estimate the abundance of several species, especially those that are individually recognizable from their natural markings, such as leopard *Panthera pardus* (e.g. Henschel & Ray 2003, Silver *et al.* 2004, Jackson *et al.* 2006, Soisalo & Cavalcanti 2006, Balme *et al.* 2009, Negrões *et al.* 2010).

The leopard is a generalist predator with a wide prey spectrum of at least 92 species in southern Africa, dominated by medium sized ungulates within the 10–40 kg range (Bailey 2005, Hayward *et al*. 2006). While leopard suffer from a high anthropogenic and natural mortality rate (Bailey 2005), they appear to be demographically viable wherever the prey base is abundant (Mondal *et al.* 2012). As a result, leopard are the most widespread large carnivore in the world (Nowell & Jackson 1996).

Despite their wide distribution, leopard were listed by the International Union for Conservation of Nature (IUCN) as one of the most threatened species in 1980 because of the international increase in demand for skins (Ray 2011). Currently leopard are listed on Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Sport hunting of leopard is allowed based on the premise that utilization does not compromise the species survival (CITES 2010). However, in several countries where leopard are hunted, no information on densities exists (Chase-Grey 2011) and their population status is usually poorly known (Packer *et al.* 2011).

Leopard are particularly sensitive to sport hunting due to the potential for infanticide by new males replacing resident individuals removed by sport hunters (Balme & Hunter 2004). The sustainability of leopard hunting has been subject to considerable debate (Chase-Grey 2011, Ray 2011), and concerns regarding leopard hunting have been raised because of the decline in harvests of leopard in Zimbabwe, and the gaps of knowledge in other southern African countries (Purchase & Mateke 2008, Balme *et al.* 2010, Packer *et al.* 2011). Similar to African lion *P. leo* (Loveridge *et al.* 2007), quantitative analysis of off-takes, and population densities are critically needed to assess the sustainability of harvest programs where leopard are hunted (Packer *et al.* 2011).

In northern Mozambique, Niassa National Reserve (NNR) is a nationally and globally important protected area for carnivore conservation (IUCN 2006). As is the case in 11 other African countries, leopard are also sport hunted in Mozambique (Balme *et al.* 2010) including inside the protected area of Niassa Reserve. Leopard are one of the most important species for the sport hunting industry, and provide an important source of revenue for NNR management (Chapter 2, SGDRN 2007).

While the protected area is large, 42,000 km^2, and supports viable populations of large carnivores and their prey (IUCN 2006, Craig 2009) 35,000 people are distributed through 40 villages inside NNR (SGDRN 2007, INE 2008a, 2008b). Little information exists on leopard population densities in miombo forest ecosystems in general (Balme *et al.* 2007) and in NNR in particular. However, evidence suggests that poaching, prey depletion, conflict with livestock, and hunting of underage individuals threaten leopard in this area (Begg & Begg 2010, Chapter 2). In the long-term, habitat loss and fragmentation induced by human population growth (INE 2008a, 2008b) are also potential threats. Thus, accurate estimates of the leopard population size in NNR are needed to ensure that legal and illegal off-takes are sustainable (Balme *et al.* 2010), to determine population status (Bailey 2005, Henschel 2008) and to monitor changes in the population over time as the result of human impacts (Ghoddousi *et al.* 2009).

In this study we provide the first population density estimates for leopard in NNR to address management concerns (SGDRN 2007) and assess the current off-take from leopard hunting. Our objectives were to: (1) estimate the leopard population densities in riparian and miombo habitats, (2) assess the turnover of leopard populations in riparian habitat over a three year period, (3) estimate the relative abundance of potential leopard prey species in both riparian and miombo habitats in NNR, and (4) assess the sustainability of leopard off-take in NNR.

3.3. Study areas

The study was conducted between 2008 and 2010 in NNR, Mozambique's largest protected area (42,000 Km^2; -12.1670S, 37.5450E), located near the border with Tanzania along the Rovuma River. NNR is one of the largest protected miombo forest ecosystems in the world, dominated by *Brachystegia* and *Julbernardia* species (Timberlake *et al.* 2004). It supports a wide range of large mammal species, in many cases comprising the largest populations in Mozambique (Agreco 2008, Craig 2009). NNR has a tropical climate and experiences a hot and dry season from mid-August through October, a hot and wet season from November through March. The mean maximum temperatures range between 18°C and 27°C, but are typically around 24°C (SGDRN 2007).

The protected area was originally proclaimed as a game reserve in 1954, and is currently managed by Sociedade para a Gestão e Desenvolvimento da Reserva do Niassa (SGDRN), a partnership between government and a private company, Investimentos Niassa Ltd. NNR is divided into 17 management units, nine hunting and six photographic tourism concessions, and two conservation units. Our study sites were located in three hunting (L3, L7 and L8) and two photographic tourism (L4 and L5S) concessions (Fig.3.1).

NNR is characterized by low productivity soils and relatively high levels of food insecurity for its human inhabitants (SGDRN 2007). Subsistence agriculture, natural resource gathering and fishing along the Lugenda River are the most important livelihoods of inhabitants (Cunliffe *et al.* 2009). The resident communities are characterized by a high degree of poverty, poor access to infrastructure and services, and high levels of conflict with wildlife, particularly African elephant *Loxodonta africana* and large carnivores (Cunliffe *et al.* 2009).

3.4. Methods

We used camera trapping data over a three year period (2008 – 2010) to obtain population density estimates for leopard and the relative abundance of their prey in riparian and miombo habitats in NNR. We classified study areas into riparian forest where more than 50% of camera trap stations were located within 100 m of the margin of perennial rivers, or miombo habitat if cameras were set in habitats dominated by *Brachystegia* and *Julbernardia* species. Four sites were chosen, two in miombo woodland (L3 study area (L3SA), and L4 study area (L4SA)) and two in riparian sites (L5S study area (L5SSA), and L7/L8 study area (L7/L8SA), Fig. 3.1). Only the riparian sites were sampled for multiple years (L7/L8SA in 2009 and 2010 and L5SSA in 2008, 2009 and 2010) and we used the same camera trap locations every year.

Two approaches were used to sample miombo and riparian study areas. To set camera stations, we deployed cameras in approximate 3 km x 3 km grids in the miombo sites, based on the smallest female leopard home range observed from radio telemetry data (10 km^2, Niassa Carnivore Project (NCP) unpublished data). In the riparian sites, no areas larger than 10 km^2 existed between cameras to ensure that no individual had zero probability of being captured. To maximize leopard captures, locations with evidence of wildlife activity and presence of water or inselberg were preferential sites for camera placement (Silver *et al.* 2004). We achieved population closure requirements (Karanth & Nichols 1998) by sampling at least 15 sites in all study areas for a minimum of 50 days. Miombo habitats were sampled in 16 sites in L4SA and 17 sites in L3SA during 60 days (26 August 2010 – 30 October 2010). Riparian habitats were sampled in 15 stations in L7/L8SA for two consecutive years (25 August 2009 – 24 October 2009, 60 days; and 01 November 2010 – 23 December 2010, 53 days) and in 15 stations in L5SSA for three consecutive years (15 August 2008 – 14 October 2008, 60 days; 25 August 2009 – 24 October 2009, 60 days; and 01 November – 23 December 2010, 53 days).

We set a pair of cameras at each station to capture both flanks of photographed individuals to allow individual identification. In 2008 only digital cameras (Cuddeback Digital Scouting Expert model, Non Typical Inc, 860 Park Lane Park Falls, WI 54552, USA) were used. In 2009 and 2010 we used one digital camera (Cuddeback Digital Scouting or Cuddeback Capture, 860 Park Lane Park Falls, WI 54552, USA) and one film camera (DeerCam DC300, 860 Park Lane Park Falls, WI 54552, USA) at each site. We attached cameras on trees30 – 45 cm above ground as recommended by Silver *et al.* (2004). Cameras functioned for 24 hours a day during the sampling periods with a 1 min delay between consecutive photos. We visited camera stations every 10 to 15 days to change batteries, memory cards, film, and to replace damaged cameras. Data loss is increasingly common in camera trapping studies (Grassman Jr. *et al.* 2005, Li *et al.* 2010), therefore for each study area we recorded the camera trapping days lost and the associated cause. When cameras were stolen, we assumed that the data loss comprised the period between our previous visit to the station and the day we detected the theft, except when camera traps were recovered and we were able to determine the last day when cameras were active.

We followed Harris *et al.* (2010) to summarize camera trapping data from riparian and miombo habitats, and determine the Relative Abundance Index (RAI) of prey species. We calculated relative abundance of prey species (capture events/100 trap-nights), as the number of independent photos of each species, which is widely used as a measure of relative abundance in camera-trapping studies (Kelly 2008). In 2009, cameras were set to record pictures by date, not by hour, and therefore we defined independent events as the number of consecutive photos of the same species photographed more than 24 hours apart, instead of the conventional number of pictures in 30 minutes interval (O'Brien *et al.* 2003). We identified mammal and avian species photographed by cameras (Maclean 1994, Skinner & Chimimba 2005) and used the literature to determine potential leopard prey in NNR (Hayward *et al.* 2006, Balme *et al.* 2007).

We identified individual leopard based on their unique spot patterns and whenever possible, determined the sex and age class of each leopard photographed through presence of external genitalia and relative body size (Chase-Grey 2011). One juvenile leopard was photographed in 2009 but was excluded from further analysis due to the low probability of young animals being photographed (Karanth & Nichols 1998). We used SPACECAP (Singh *et al.* 2010a) to estimate population density of leopard in all study areas. This software uses closed-model capture-recapture sampling based on photographic captures. It implements Spatially Explicit Capture-Recapture (SECR) models developed by Royle *et al.* (2009), and minimizes the risk of inflated density estimates in the analysis of our camera trapping data

We created home range centres using ArcGIS 9.3 (Environmental Systems Research Institute, ESRI; Redlands, CA:). We drew a polygon enclosing the outermost camera trap stations and added a large enough buffer to ensure that no individual outside of the buffered area had any probability of being captured by cameras during the survey. A grid representing leopard home range centres was created for each site (Fig. 3.2). To create a state-space file we examined the suitability of habitats of each potential home range center. For each location, we indicated with a 1 when the habitat was suitable for leopard or 0 if it was unsuitable habitat, such as NNR headquarters.

We used Kruskal-Wallis non-parametric ANOVA to compare the RAI of leopard potential prey in riparian and miombo habitats. The Post-hoc Mann-Whitney test was used to compare pairs of habitats when there were significant differences on the previous test. Furthermore we used the spearman correlation to test the relationship between the leopard population density and the relative abundance of prey. All statistical analyses were performed in SPSS version 19.0 (IBM SPSS).

We obtained sport hunting data on leopard quotas and safaris (2004–2010, SGDRN records), and the estimated ages of individuals harvested (2004–2010, NCP records), in NNR hunting concessions. The skull of each individual was examined before it left the reserve, and placed into one of the three age classes (≤ 2 years, 3-4 years, and ≥ 4 years) based on the tooth wear (Stander 1997). Leopard quotas were analyzed based on the size of hunting blocks where they were hunted to determine the legal hunting intensity (average number of leopard hunted per 100 km^2, Packer *et al.* 2011) over a seven year period (2004–2010). We based the illegal hunting intensity on the number of radio-collared leopard poached and size of the block where poaching occurred. We used an off-take range of 0.1–0.3 leopard/100 km^2 (as per Packer *et al*. 2011) to assess the sustainability of leopard sport hunting from 2004 to 2010 in NNR.

The vegetation of individual NNR hunting concessions included both riparian and miombo habitat types (SGDRN 2007). We therefore used the hunting locations of individual leopard trophies (NCP records) to determine the proportion of leopard taken in the different habitat types. To calculate the density of male leopard we used the sex ratio of leopard identified in miombo and riparian study areas and the range of leopard population density estimated for the major habitat type where leopard were hunted. We determined the hunting intensity on the density using the formula: hunting intensity/density (100 km^2)*100 (Ray 2011).

3.5. Results

Over three years, we identified 48 adult individual leopard (21 male, 25 female and two unknown sex) from 64 independent photo-captures at 49% (n = 31) of the camera trapping stations (Table 3.1). Although we could not determine the sex of two individuals captured, the configuration of rosettes on their pelage allowed us to differentiate them from identified individuals of known sex. None of the individual leopard identified in riparian study site L5SSA were recorded in riparian study site L7/L8SA, despite the close proximity of the two sites on opposite banks of the Lugenda River and concurrent surveying. This suggests that leopard did not cross the Lugenda River during the study period. A total of 285 camera trapping days were lost in 2009 and 2010 due to electronic problems, theft and damage to cameras by elephants and fires, but these days were accounted for in SPACECAP when calculating leopard densities.

Differences in leopard population densities estimated by SPACECAP were observed between habitat types, with almost double the density in riparian areas compared to miombo forests (Fig. 3.3). For riparian areas, where multiple years of camera trapping data were collected, leopard densities were relatively stable between 2008 and 2009 (mean ± SD; 9.92 ± 2.7 leopard/100 km^2 to 12.65 ± 3.7 leopard/100 km^2). In 2010 the population density decreased to 3.47 ± 1.27 leopard/100 km^2 in L5SSA and to 7.54 ± 2.33 leopard/100 km^2 in L7/L8SA. However, the decrease was only significantly in L5SSA compared to previous years ($p < 0.05$, Fig. 3.3). The leopard population densities) in L5SSA a non sport hunted area in 2008 (9.92 ± 2.7 leopard/100 km^2 and) in 2009 (10.01 ± 2.9 leopard/100 km ^{2}were not significantly different ($p > 0.05$) from the population density

(12.65 ± 3.7 leopard/100 km^2) in L7/L8SA a sport hunted area in 2009. The leopard population densities in riparian sites (sport hunted and non sport hunted) in 2010 were also not significantly different. Similarly, in miombo, the leopard population density did not differ significantly between the hunting site (4.31 ± 1.34 leopard/100 km^2) and the photographic site ($p > 0.05$). There were therefore no significant differences in the leopard densities in hunted and non hunted areas in both miombo and riparian study sites. Although, leopard population densities were lower in miombo sites than in riparian sites, densities were only significantly lower ($p < 0.05$, Fig. 3.3) in L4SA (2.18 ± 0.37 leopard/100 km^2) compared to riparian sites, in all years except in L5SSA in 2010.

Of the 14 individuals photographed in 2008 in L5SSA, seven were recaptured in 2009 but no individual was recaptured in 2010. Similarly in L7/L8SA camera traps only photo-captured new individuals during 2010. The reduction in photo-captures as well as the reduction in known animal photo-captured indicates high rates of turnover in the leopard populations in riparian sites caused by human actions. Three radio-collared female leopard photographed in L5SSA in 2008 and 2009, were killed in wire snares set for bushmeat. The outcome of other identified leopard is unknown.

Of the 44 species, excluding leopard, photographed by camera traps over 6320 camera trapping days in riparian and miombo habitats, 18 were potential leopard prey species (Table 3.2). These comprised ungulates (53%), primates (24%), rodents (12%), and birds (12%). The total Relative Abundance Index (RAI) of potential prey species in riparian sites was at least double (39.5 in 2009 and 52 in 2010) that of miombo (15.6). The photo-captures of impala represented 38% and 58% of riparian ungulate RAI in 2009 and 2010. The Kruskal-Wallis test indicated significant differences in the relative abundance of impala *Aepyceros melampus* between study areas ($H(2) = 35.94$, $p < 0.001$). The Mann-Whitney U Post-hoc test showed that the RAI of impala was significantly higher in riparian sites in 2009 ($U = 198.5$, $p < 0.001$) and in 2010 ($U = 95$, $p < 0.001$) than in miombo sites. The test also indicated that the relative abundance of impala increased significantly within riparian sites between the dry season in 2009 and the beginning of wet season, around November, in 2010 ($U = 242$, $p = 0.005$), when female impala had their lambs (A. Jorge personal observation; Table 3.2).

In miombo woodland, common duiker *Sylvicapra grimmia* had the highest relative abundance (43% of the ungulate RAI). The relative abundance of common duiker in miombo sites was significantly higher ($p < 0.001$) than in riparian sites in 2009 and 2010.

Apart from ungulates, only yellow baboon had significant differences in the RAI between study sites. The relative abundance of rodents and birds species did not differ significantly between riparian and miombo sites (Table 3.2). There was no significant correlation between leopard population density and relative abundance of prey species ($r > 0.135$, $p > 0.05$).

Leopard hunted by sport hunters over a seven year period were grouped into three age classes based on tooth wear (≤ 2 years: 37%; 3–4 years: 48%; ≥ 4 years: 15%; Fig. 3.4). In general, the percentage of different leopard age classes hunted was relatively constant across different years. At least 41% of trophies targeted by sport hunters in all years were 3-4 years old. Young individuals (≤ 2 years) were the second most hunted age class in NNR, with the highest percentage in 2010, when they comprised 42% of harvests. The percentage of individuals ≥ 4 years old was lower than all other classes, except in 2005 when they were similar to the percentage of young individuals hunted.

During seven years, NNR operators hunted 106 leopard out of the 163 leopard trophies available on quota. The annual leopard hunting off-take per concession was an average of 64% of the approved quota (range: 0–112%). On average, two leopard were hunted per concession per year (range: 0–5 leopard), except in 2006 and 2007 when concessionaires hunted an average of three individuals. The resulting mean (±SE) hunting intensity for leopard was 0.068 ± 0.01 leopard/100 km^2 (range: 0–0.234 leopard/100 km^2). The highest hunting intensity (0.234 leopard/100 km^2) was observed in 2010 when an operator used all (five) leopard on quota. Although the overall annual leopard off-take has increased from 11 leopard in 2004 to 19 leopard in 2010 the total area available for leopard hunting has also increased (from 20,370 km^2 to 27,981 km^2) within seven year period as more hunting concessions have become operational. The NNR off-take was ecologically sustainable based on the limits from Packer *et al.* (2011) (Fig. 3.3).

Based on the analysis of trophy locations between 2007 and 2011, hunting operators concentrated their leopard hunting activities in riparian forest rather than in miombo woodland. The majority of leopards were hunted on baits close to the rivers (< 1 km) with a high percentage of leopard taken from riparian habitats (66%, n = 43). The remainder were hunted relatively far from rivers (1–2 km, 17%, n = 11; and > 2 km, 17%, n = 11). The off-take from the riparian safaris was equivalent to 2–

7% of adult male leopard population in L7/L8 concession based on the range of the riparian leopard population density (3.47 ± 1.27 leopard/100km^2 to 12.65 ± 3.70 leopard/100 km^2) determined from the camera trapping data. In the non commercial hunting concession, the lower hunting intensity resulted in smaller proportion (1–3%) of the adult male leopard harvested (Table 3.1).

The illegal off-take of at least three female leopard in L5S was equal or greater than the numbers sport hunted in seven of the nine hunting concessions in 2010. The intensity from this illegal hunting (3 leopard/800 km^2) was also higher than the hunting intensity in all hunting concessions in all years in NNR (Fig. 3.4). Furthermore, this off-take was above the upper limit of sustainability recommended by Packet *et al.* (2011). Based on the range of the riparian leopard population density (3.47 ± 1.27–12.65 ± 3.70 leopard/100 km^2), poachers removed an equivalent of 6–22% of female adult population (Table 3.1). This clearly indicates that the numbers of leopard killed by sport hunters combined with poachers' off-take, are not ecological sustainable in NNR.

3.6. Discussion

Our results represent an important step towards understanding the status of leopard populations in Niassa National Reserve (NNR) and the pressure they face from sport hunting and poaching.
Previous studies recognized miombo ecosystem as an important habitat for leopard conservation (Balme *et al*. 2007), and we provide further evidence on the relevance of this ecosystem for leopard, and also of the importance of NNR for large carnivore conservation at national and regional levels. Our density estimates for leopard in miombo forest and riparian habitats are comparable with those found in Phinda Private Game Reserve and Zululand Rhino Reserve, South Africa, and lowland rainforest areas of Lopé and Ivindo National Parks, Gabon (2.7–12.1 leopard/100 km^2, Balme *et al.* 2007, Chapman & Balme 2010, Henschel *et al.* 2011). Our estimates are lower than the 20 leopard/100 km^2 in Soutpansberg mountains, South Africa (Chase-Grey 2011) but higher than the 1–3.6 leopard/100 km^2 in Namibia (Stein *et al.* 2011). We acknowledge the risks of comparing population densities estimates obtained using different methods (Gerber *et al.* 2011), particularly when there is a lack of consensus on the most accurate way to determine the size of area effectively trapped (Soisalo & Calvacanti 2006, Balme *et al.* 2007, Gerber *et al.* 2011). However, by using Spatial Explicit Capture Recapture models (Efford *et al.* 2009, Royle *at al.* 2009) we did not make subjective decisions regarding the Effective Trapped Area, therefore our densities estimates can be

comparable with future leopard population density estimates in NNR, and elsewhere, using a similar approach.

We also acknowledge the potential impact of human and natural effects on our camera trapping data. In 2010 we might have photo-captured fewer leopard because we surveyed riparian study areas towards the end of dry season and beginning of the dry season whereas in the previous years riparian habitat was sampled only in the dry season. In addition, failure and theft of camera-traps (Grassman Jr *et al.* 2005, Dajun *et al.* 2006, Singh *et al.* 2010b) in 2010, and our determination of Relative Abundance Index (RAI) of prey (O'Brien *et al.* 2003) may be potential problems in our analysis of species abundances.

Densities of large carnivores may vary between sites because of vegetation type, prey availability and hunting pressure (Koehler & Pierce 2003, Chapron *et al.* 2008). The riparian study areas are clearly more productive than miombo areas (Natta *et al.* 2003, WWF *et al.* 2010), with higher RAI particularly of the ungulates in the preferred body mass range of leopard prey, such as impala (Hayward *et al.* 2006). Similar to tiger *Panthera tigris* (Karanth & Sunquist 1995), the higher densities of leopard in riparian habitats compared to miombo habitats are likely to be related to the increased abundance of their potential prey species in riparian habitats (Hayward *et al.* 2006, Balme *et al.* 2007).

The off-take from illegal hunting is additive to sport hunting off-take, and overall leopard mortality might not be sustainable in NNR. Although sample sizes are small, no significant differences in leopard densities between sites in hunting and photographic concession were observed, which may suggest that the numbers of leopard sport hunted have limited impact on the populations compared to other anthropogenic actions. Poaching exerts high pressure on large carnivore populations (Persson *et al.* 2009, Liberg *et al.* 2011) and is not selective in terms of sex and age of individuals targeted (Goodrich *et al.* 2008, 2010). In NNR, no sport hunting occurred in L5S (SGDRN 2007) and it is unlikely that individual leopard moved between L7/L8 hunting and L5S photographic concessions during the study period. Therefore, the observed turnover and the decline in the leopard populations in L5S were most likely caused by poaching as was the case for the three female leopard confirmed killed in bushmeat snares. Poaching of female leopard is a serious conservation concern because it has consequences beyond one individual (Goodrich *et al*. 2010) and cubs may also die after the poaching of adult females as demonstrated by Goodrich *et al.* (2008). Thus poaching is likely to be a serious threat to leopard conservation in NNR and elsewhere. Hence, it is critical to quantify and include effects of poaching in management policies of leopard and other large carnivore (Hotte 2006, Karanth *et al.* 2011, Liberg *et al.* 2011). Our results showed no significant correlation between leopard density and prey density contrary to other studies where the density of large carnivore populations was positively correlated with prey abundance in areas with low poaching incidence (Kiffner *et al.* 2009). This may suggest that leopard densities in NNR are currently constrained by the high levels of poaching off-take rather than prey.

The overall leopard legal hunting intensity for NNR is within the sustainable limits recommended by Packer *et al.* (2011), although the sustainability of leopard hunting might vary across individual hunting concessions. The sustainable off-take of the adult male leopard population is 3.8% (Caro *et al.* 2009) and only males are allowed to be sport hunted in NNR (SGDRN 2010). However, according to Caro *et al.* (2009) the sustainable leopard off-take is below the 3.8% when adult female leopard and young individuals are hunted. This is likely to be exacerbated by the fact that operators are not hunting across all of their blocks, but a portion thereof (Begg & Begg 2008). Packer *et al.* (2009) suggest around seven years as a safe age for hunting leopard to minimize infanticide, however, few leopard older than 4 years of age are currently being taken in NNR. The potential negative effect of infanticide caused by sport hunting in NNR even though off take appears to be sustainable should not be underestimated (Balme *et al.* 2010, Ray 2011).

Different to lion (Whitman & Packer 2007), implementing an age based sport hunting system for leopard is complicated by the lack of obvious visual aging cues that sport hunters can use to age the animal before it is shot (Packer *et al.* 2011). Until age based visual criteria have been validated for leopard, an alternative would be to implement a site rotation system within each concession, with no hunting zones declared around bait sites for 2-3 years where leopard had been previously hunted. This would minimize the effects of infanticide and destabilization of the leopard population (Ray 2011). Furthermore, the proposed sustainable harvest levels for Zambia (1–2 leopard/1,000 km^2, Ray 2011) appears more realistic for NNR, compared to those from Tanzania (1–3 leopard/1,000 km^2, Packer *et al.* 2011) because of human population pressures and the existence of substantial illegal off-take in the reserve.

Similarly to Congo Basin rainforest (Henschel *et al.* 2011), the growing human population in NNR (INE 2008a, 2008b), the improvement of road access inside and around the reserve, and the lack of fences to regulate access of people to the resources (Geddes 2006, SGDRN 2007), are likely causes for lower densities of leopard in some parts of NNR that are heavily utilized for resource extraction. It is therefore important to zone areas with rural communities inside the reserve (Ngoprasert *et al.* 2007, Ordoñez 2007) to limit anthropogenic activities and ensure conservation of large carnivores (Woodroffe & Ginsberg 1998, Balme *et al.* 2009).

Future decisions regarding any increase on the current leopard quota should be based on effective monitoring of the leopard populations in NNR, because they are not only sport hunted but are also illegally hunted inside the protected area. Camera trapping studies will be useful to monitor human-mediated removal of leopard (Woodroffe & Ginsberg 1998, Henschel & Ray 2003) to determine how seasonal relative abundance of prey affects leopard population dynamics in riparian habitats, to assess leopard densities in individual concessions (Balme *et al.* 2009, Henschel *et al.* 2011), and to measure their long-term response to interventions such as Payments to Encourage Carnivore Coexistence (PEC) in NNR (Chapter 2). The success and sustainability of NNR leopard hunting in the future will be challenged by the ability of the stakeholders of the hunting industry to control and mitigate leopard mortality from poaching and conflict with livestock (Balme *et al.* 2010).

3.7. Conclusion

NNR leopard densities were comparable with other parts in Africa, which suggest that the current leopard conservation status is good, but it is clearly threatened by anthropogenic activities. The decline of leopard population densities in riparian zones and illegal off-take indicates that declaring leopard protected by the law is not enough to ensure its protection, even inside a national reserve. Long-term solutions for reducing poaching of leopard and other species are clearly needed and will require proactive action at the reserve level, and most importantly, will require the support of local communities sharing land with leopard and other large carnivores (Chapter 2).

Sport hunting is a relevant source of revenue for conservation of leopard and other large carnivores in NNR (Chapter 2), however the persistence of leopard populations in the wild will depend on the ability to maintain the concessionaires off-take within sustainable limits based on the number and age of the individuals taken, and on controlling the illegal off-take. Further camera trapping studies in NNR might provide useful information for management of leopard and other large carnivores in NNR and can be used to monitor leopard conservation status over time.

Table 3.1. Leopard sex ratio and male density in riparian and miombo habitats across all sampling years. Also shown are the hunting intensity and percentage off-take on male and female density in hunting and photographic concessions.

Year	Concession	Sex ratio (male/female)[a]	Mean density males/100km^2	Hunting intensity (leopard/100 km^2)[d]	Leopard off-take /100 km^2[e]
2009	L7/L8	1:3 (33%) [b]	1.15–4.17	0.077	2–7%
2010	L7/L8	3:2 (60%)	2.08–7.59	0.140	2–7%
2010	L5S	1:1 (50%)	1.74–6.33	0.375	6–22%
2010	L3	1:1 (50%)[c]	1.74–6.33	0.038	1–2%

[a]based on the sex of individual leopard photographed in 2008-2010 camera trapping study in NNR.

[b]two individuals of unknown sex.

[c]one individual of unknown sex.

[d]based on the hunting intensity observed in each concession.

[e]off-take of adult leopard male, except for L5S where it indicates the female off-take by poachers.

Table 3.2. Average (±SE) values for Relative Abundance Indices (RAI, independent photos/100 trap nights) for potential prey species of leopard photographed at camera traps station in riparian and miombo habitats between 2009 and 2010. Also shown the *p* values for the comparison of RAI from Kruskal-Wallis test with bold letters showing significance from pairwise comparisons between riparian sites (2009 and 2010) and miombo study area.

Species*	**Riparian**		**Miombo**	***p***
	2009	**2010**	**2010**	**value**
Ungulates				
Impala *Aepyceros melampus*	0.36 ± 0.028(**a**)	0.80 ± 0.04(**b**)	0.04 ± 0.01(**c**)	< 0.001
Bushbuck *Tragelaphus scriptus*	0.29 ± 0.02(**a**)	0.16 ± 0.06(**a**)	0.11 ± 0.08(**b**)	0.032
Common duiker *Sylvicapra grimmia*	0.04 ± 0.01(**a**)	0.05 ± 0.011(**a**)	0.47 ± 0.02(**b**)	<0.001
Antbear *Orycteropus afer*	0.002 ± 0.004(**a**)	0.018 ± 0.008(**b**)	0.09 ± 0.02(**b**)	0.021
Warthog *Phacochoerus africanus*	0.032 ± 0.013(**a**)	0.112 ± 0.015(**b**)	0.074 ± 0.011(**b**)	0.012
Klipspringer *Oreotragus oreotragus*	0.013 ± 0.013(**a**)	0.007 ± 0.005(**a**)	0.020 ± 0.009(**a**)	0.998
Red duiker *Cephalophus natalensis*	0.002 ± 0.004(**a**)	0(**a**)	0.020 ± 0.008(**a**)	0.078
Reedbuck *Redunca arundinum*	0(**a**)	0.002 ± 0.004(**a**)	0.015 ± 0.009(**a**)	0.623
Bushpig *Potamochoerus larvatus*	0.002 ± 0.003(**a**)	0.011 ± 0.005(**a**)	0.037 ± 0.01(**a**)	0.133
Rodents				
Cape porcupine *Hystrix africaeaustralis*	0.030 ± 0.008(**a**)	0.011 ± 0.006(**a**)	0.080 ± 0.012(**a**)	0.053
Scrub hare *Lepus saxatilis*	0.028 ± 0.009(**a**)	0.011 ± 0.007(**a**)	0.013 ± 0.008(**a**)	0.203
Muridae species	0.017 ± 0.009(**a**)	0(**a**)	0(**a**)	0.204
Primates				
Yellow baboon *Papio cynocephalus*	0.119 ± 0.018(**a**)	0.227 ± 0.019(**b**)	0.094 ± 0.018(**a**)	0.032
Vervet monkey *Chlorocebus aethiops*	0.062 ± 0.013(**a**)	0.083 ± 0.014(**a**)	0(**b**)	0.002
Samango monkey *Cercopithecus mitis*	0.019 ± 0.009(**a**)	0.004 ± 0.004(**a**)	0(**a**)	0.196
Greater galago *Otolemur crassicaudatus*	0.004 ± 0.004(**a**)	0.009 ± 0.007(**a**)	0(**a**)	0.332
Birds				
Helmeted Guineafowl *Numida meleagris*	0.074 ± 0.015(**a**)	0.031 ± 0.012(**a**)	0.020 ± 0.010(**a**)	0.640
Francolin *Francolinus afer*	0.015 ± 0.009(**a**)	0.007 ± 0.006(**a**)	0(**a**)	0.202

*The photo-capture of semi-arboreal leopard prey species were interpreted as a measure of their terrestrial availability.

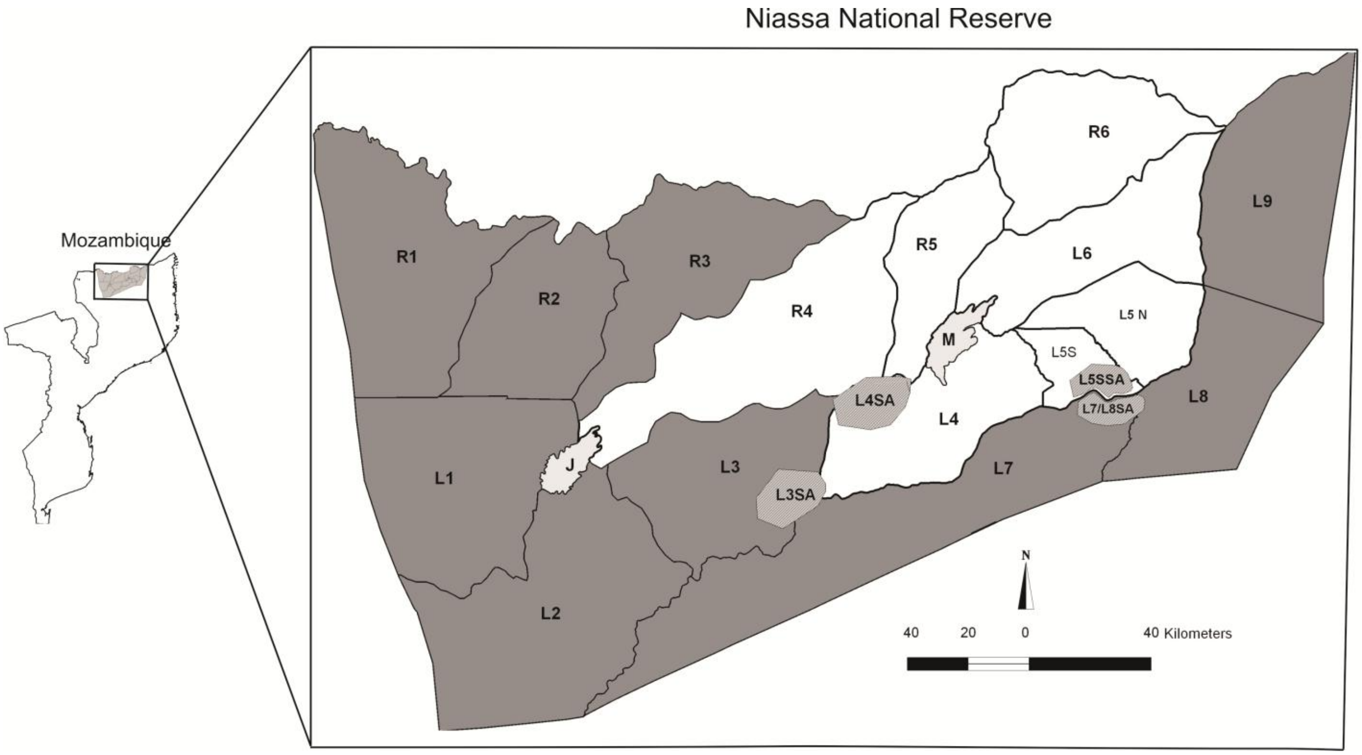

Figure 3.1. The location of our four study areas (L3SA, L4SA, L5SSA and L7/L8SA) in relation to the nine hunting (dark grey) and seven photographic tourism (white) concessions, and two resource conservation (J and M, light grey) management units of NNR.

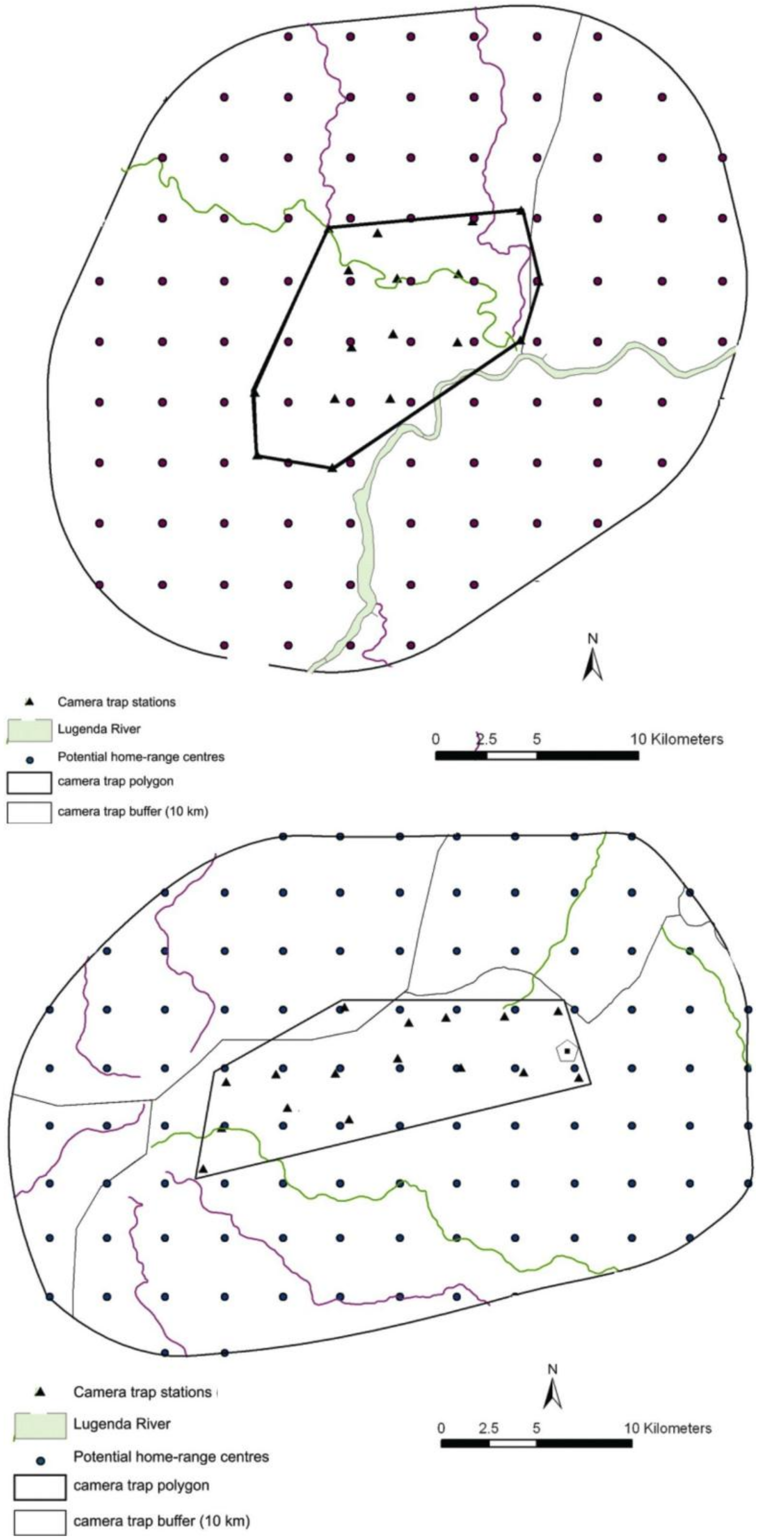

Figure 3.2. Map illustrating potential leopard home range centres in miombo L3 and L4 study areas.

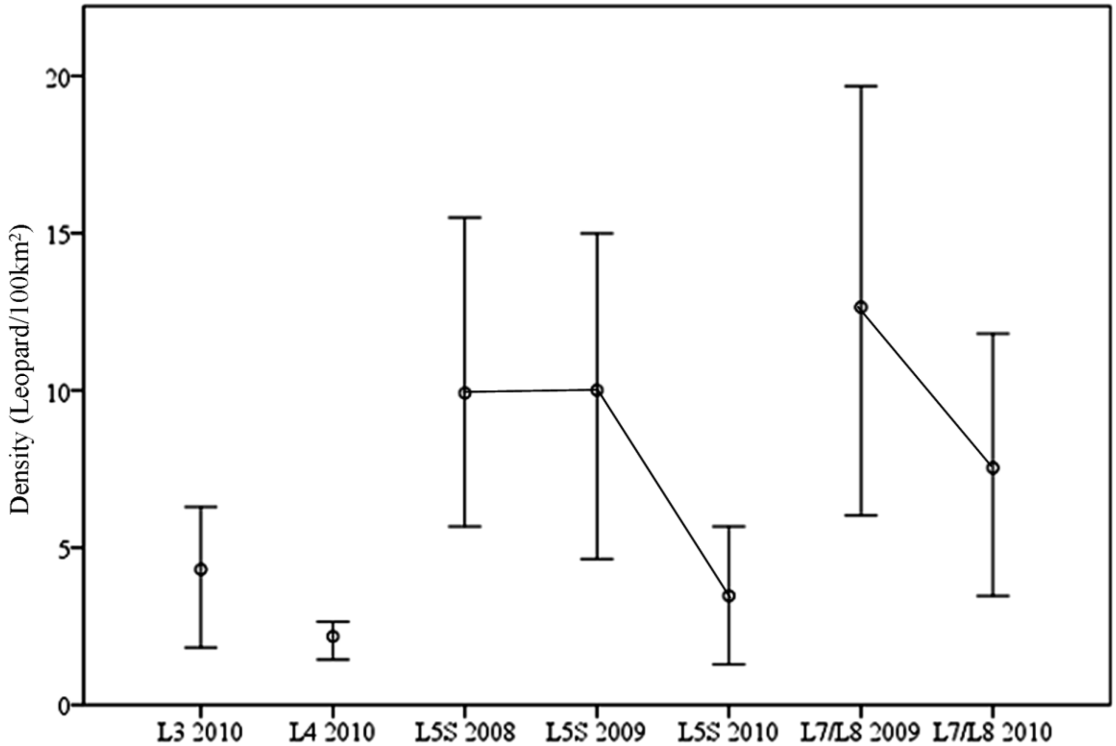

Figure 3.3. Mean leopard population density and 95% confidence limit estimated through SPACECAP, among riparian (L3 and L4) and miombo sites (L5S and L7/L8) in NNR, 2008-2010.

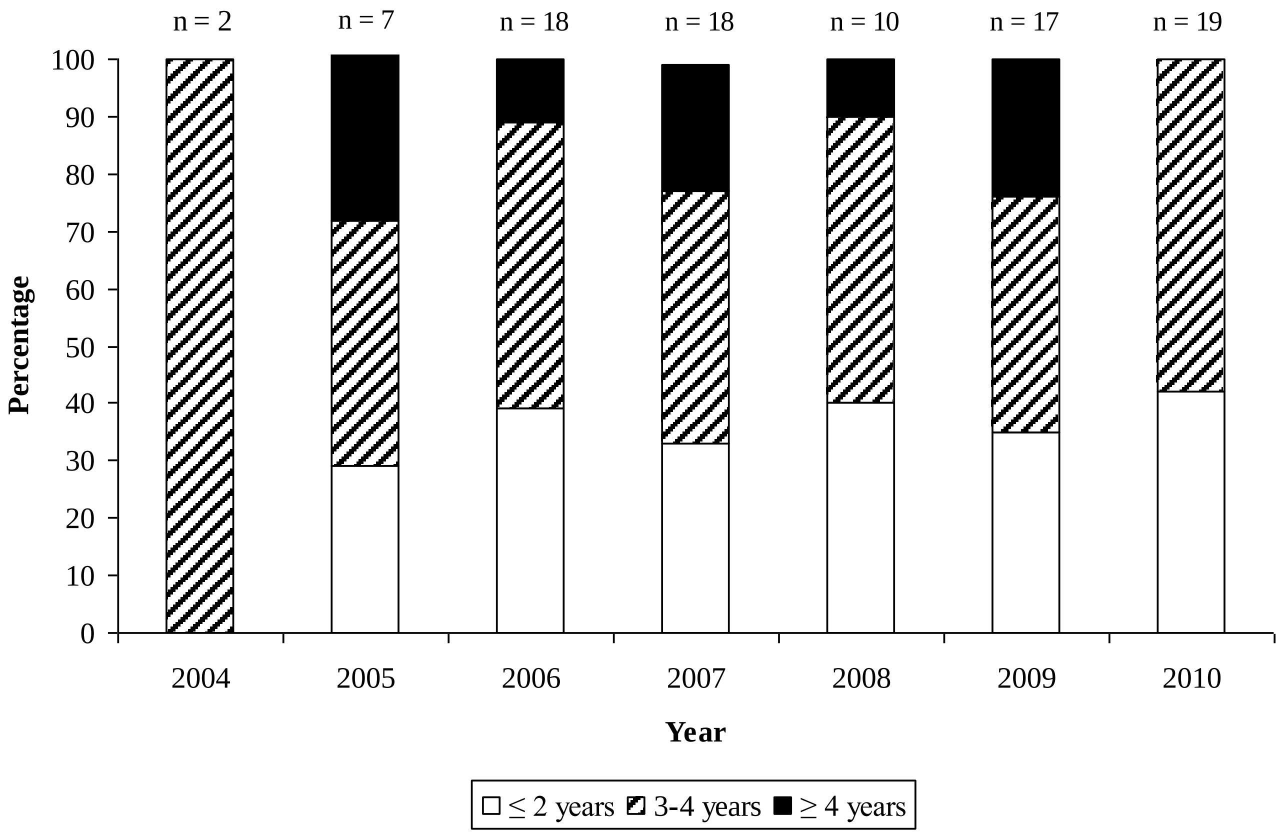

Figure 3.4. Age classes of leopard trophies (n = 91) hunted in the NNR hunting concessions, 2004-2009. The 2004-2009 data is based on information from Begg & Begg 2009, with permission.

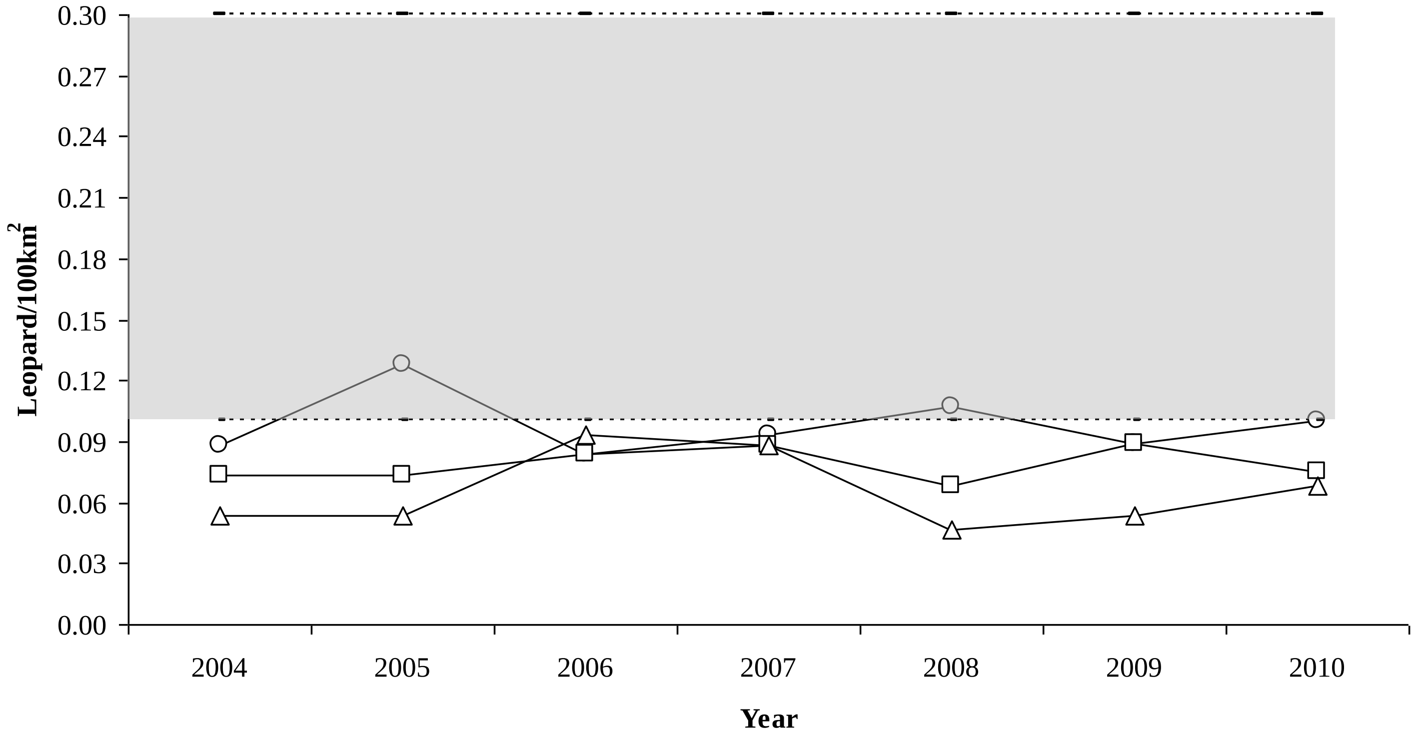

Figure 3.5. Leopard quotas approved (open circle) and purchased (open square), and off-take (open triangle) in the NNR hunting concessions from 2004 - 2010. Lower and upper limits of sustainable leopard off-take for Tanzania (as per Packer *et al.* 2011) are indicated with the grey band.

CHAPTER 4

Conclusions and management recommendations

The price of trophies in the hunting industry is dependent on a species' International Union for Conservation of Nature (IUCN) threat status, and changes in the threat status are often reflected in changes in hunting pressure (Palazy *et al.* 2011, Prescott *et al.* 2011). Similar to the African lion *Panthera leo* in Zimbabwe, where the ban imposed in 2005 caused the trophy price to rise from $4,000 to $30,000 (Johnson *et al.* 2010), the decline of leopard *P. pardus* globally and their increased conservation status might also raise their trophy prices in Niassa National Reserve (NNR) and elsewhere. While NNR concessionaires might derive more revenue per individual leopard hunted if the trophy value increases, this can also act as a strong incentive for illegal hunting (Palazy *et al.* 2012). Consequently, operators would have to invest more resources to protect the species (Palazy *et al.* 2012). It should be in the interests of all stakeholders of the sport hunting industry in NNR and in Mozambique to prevent future declines in the leopard to avoid compromising both ecological and economic sustainability of leopard hunting.

Evidence suggest that, in southern Africa, the hunting industry relies on elephant *Loxodonta africana* (Botswana Wildlife Management Association 2001, Booth 2005), buffalo *Syncerus caffer* (Baldus & Cauldwell 2004), lion (Lindsey *et al.* 2012) and plain's-game safaris (Damm 2005, Booth 2009), rather than leopard, to generate revenues. However, my results in Chapter 2 demonstrated that the NNR hunting industry depends mostly on leopard to generate revenues to stakeholders. This is possibly because elephant (Cumming & Jones 2005) and plain's-game (Lindsey *et al.* 2007c) are relatively more abundant in other countries compared to NNR. In addition, in NNR leopard are widely distributed across the reserve compared to the limited distribution of elephant which are concentrated in the eastern section of the protected area (Craig 2009), and lion which have a low density overall. In addition, within NNR elephant and lion hunts are subject to stricter rules than is the case for leopard hunts (SGDRN 2006a, 2006b), with economic penalties to operators when young elephant or lion are harvested. Leopard are highly preferred by hunters from the USA (Lindsey *et al.* 2006), the most important source of clientele for NNR operators, possibly because hunters cannot import NNR elephant trophies into the USA due to US Fish and Wildlife Service restrictions (Booth 2009). This suggests that the economic sustainability of hunting businesses depends not only on the availability of high valuable species on quota, but also on the context in which the industry operates, and, most important, on the availability of individuals that can be hunted sustainably.

My results also indicated that revenues from leopard hunting are largely captured at international and national levels, with limited amounts accruing to NNR communities (Chapter 2). This problem has previously been raised by other researchers (Dickman *et al.* 2011, Palazy *et al.* 2011) in other areas, but limited quantitative analysis of revenue distribution was presented. At the national level, NNR is one of few protected areas generating revenues for government and NNR leopard revenues represent approximately 50% of the total profits accrued to the Ministry of Tourism (MITUR) from national sale of leopard trophies' licenses ($25,000; Magane *et al.* 2009).

The revenues from leopard hunting in NNR may contribute to their protection in hunting concessions because of their importance there. However in NNR, the large areas requiring protection combined with the low capacity of anti-poaching teams to effectively patrol the reserve (SGDRN 2007) contributes to relatively low e opportunity costs for being arrested for poaching. In other areas, the potential costs of getting caught are high and these opportunity costs have been shown to significantly reduce illegal hunting (Messer 2000, 2010). Nonetheless, patrols by operators' scouts in NNR resulted in the apprehension of 24 poachers, confiscation of three illegal skins of leopard and lion, seizure of eight firearms, and removal of more than 1,220 snare lines between 2004 and 2010 (SGDRN unpublished data). However, because concessionaires differ in the amount of sport hunting benefits generated and differ in the proportion of their revenues reinvested on anti-poaching (vehicle, human resources, and patrol effort), the level of protection to leopard and other species is likely to vary among concessions. Furthermore, operators differ in terms of their presence in the block throughout the year, and are absent during the wet season (Mayaka *et al.* 2005).

There are problems with distributing revenues from leopard sport hunting equitably among communities and it is not clear to communities that the revenue distributed comes specifically from leopard or other wildlife species. Although the overall community hunting benefits are comparable to funds allocated by the government to development initiatives at district level ($233,333; Conselho de Ministros 2009), the leopard benefits represent a very small portion of the government fund (3%). Moreover, when distributed per household (< $1), leopard revenues are clearly inferior to the profits that an individual household could make from poaching and the sale of leopard skins and are also lower than the costs of leopard depredation on goats. Furthermore, there is little linkage between the revenues from leopard hunting and the conservation actions desired from communities by the management authority (SGDRN 2007, Cunliffe *et al.* 2009).

There is a strong argument to introduce and implement a Payments to Encourage Coexistence (PEC) scheme targeting conservation of leopard and other large carnivores in NNR, based on their threat to communities and the need to overcome their costs at the local scale. The value of leopard in both consumptive and non consumptive tourism is widely recognized (Lindsey *et al.* 2006, 2007c). The challenge of turning this recognition into effective persistence of leopard at local scale could be partially achieved through tourists' conditional payments to local communities (Naidoo *et al.* 2011). Payments to local communities to coexist with leopard have the potential to make carnivore conservation a viable land use option to NNR villagers, and to benefit the general public at national and global scales. However, PEC is mostly like to benefit communities when implemented on lands poor for agriculture, but with high potential for conservation (Bulte *et al.* 2008). Furthermore, payments need to be greater than the opportunity costs that villagers incur by coexisting with carnivores (Nelson 2009).

To operate PEC, it is essential that participants accept that conservation of leopard and other large carnivores is based on negotiated payments to local communities and that the long-term financing mechanisms are available (Pagiola *et al.* 2005, Wunder 2007, Wunder *et al.* 2008, Nelson 2009). Tourism is a growing sector in Mozambique and elsewhere (MITUR 2003), therefore photographic and sport hunting operations in NNR might provide significant revenues for long-term financing of PEC in the reserve.

Although photographic tourism and sport hunting may seem mutually exclusive activities (Elliott & Mwangi 1998), their simultaneous occurrence inside one hunting concession in NNR suggests that they can coexist in larger areas (Novelli *et al.* 2006). The remoteness of NNR, lack of tourism infrastructure, the high operational costs, and lack of habituated wildlife, have contributed to the low development of photographic tourism in the reserve (SGDRN 2007, A. Jorge & C. Begg unpublished data). But interestingly, several operators (n = 3) running concessions for more than four years in NNR are already including, photographic tourism as an alternative product to sport hunting (SGDRN unpublished data). While this might fit with one of the goals of SGDRN for operators to gradually shift their hunting operations into photographic tourism infrastructure improves inside the concessions (SGDRN 2007), it will take time before NNR fully develops its photographic tourism potential and competes with other international tourism destinations in Africa.

In 2010, community tourism benefit would increase by approximately 50% if PEC targeting leopard was implemented in NNR (Table 4.1). Most revenues would be derived from community leopard hunting and plain’s-game safaris, but households could also derive direct income from employment and sales of goat baits to operators. Extending the use of baits to photographic operations can increase chances for clients observing leopard in photographic safaris (Stein *et al.* 2010), and consequently could raise household income. However, despite its potential to contribute to household economies, potential concerns from animal welfare organizations and other groups regarding the use of bait should be considered (Lindsey *et al.* 2007a).

Diversification of ecological services is critical for success of Payment for Ecosystem Services (PES) initiatives (Naidoo *et al.* 2011). Because of the relatively large human population inside NNR, a PEC scheme funded solely by tourism might not provide enough revenue to off-set potential gains from poaching, and thus may not motivate villagers to voluntarily conserve leopard to protect their income. Certifying tourism operations in NNR should therefore be considered an option to increase revenues for communities and conservation (Packer 2005, Dreike 2007, Child & Wall 2009). In addition, by conserving habitats for large carnivores and other wildlife species, local communities might reduce activities, such as slash and burn, with adverse impact on the environment, making them eligible to other PES initiatives such as Reducing Emissions from Deforestation and Degradation (REDD) programs (Engel *et al.* 2008). Finally, carnivore conservation in NNR can also be financed through donor funding considered as a transfer of funds from beneficiaries of carnivore existence values (in the developed world) to villagers who suffer their existence (MacDonald & Sillero-Zubiri 2002, Naidoo *et al.* 2011).

Experiences from community based conservation projects indicate that distribution of tourism revenues can be problematic and does not always improve conservation (Johannesen & Skonhoft 2005). However the involvement of relatively poor communities on carnivore conservation in PES scheme in India (Mishra *et al.* 2003) and Namibia (Naidoo *et al.* 2011), and in ‘PES-like’ scheme in Zimbabwe (Frost & Bond 2008) are encouraging examples for PEC implementation in NNR. Nonetheless, prior to implement PEC in NNR it is critical to conduct further research and learn from failures experienced in other conservation projects (see Webber *et al.* 2007, Redford *et al.* 2011).

In Chapter 3, I demonstrated that NNR is an important habitat for leopard conservation where the status of leopard can be regarded as secure under the existing conditions and the numbers of individuals harvested by concessionaires are sustainable. However, while it has long been assumed that leopard can persist under anthropogenic threats due to their great adaptability (Henschel *et al.* 2008), my results suggest that humans are severely affecting dynamics of the leopard population by directly removing individual leopard through snaring both of individual leopards and their prey (Henschel *et al.* 2008, 2011). Ongoing and increasing pressures could compromise the secure status of leopard in NNR in the future. Furthermore, the limited power of the CITES at national and local levels, and the inefficiency of domestic legislation to deter poachers (Adams *et al.* 2009) translates into low protection of leopard (Balme 2009) in NNR at present.

In addition, while the off-take of leopard for sport hunting is not necessarily incompatible with conservation goals (Macdonald & Sillero-Zubiri 2002), large carnivores can only tolerate low levels of hunting (Balme *et al.* 2010, Packer *et al.* 2010). Although my results indicate that the concessionaires' off-take is sustainable I terms of numbers of leopard hunted, I caution readers against concluding that leopard can be hunted sustainably elsewhere and indefinitely. In Norway, Linnell *et al.* (2010) demonstrated that sustainable management of the harvest of large carnivore populations is possible, however it can be challenging even when monitoring data are available. In Africa, sport hunting has been the main cause of population declines in lion and leopard through subsequent infanticide in some areas (Whitman *et al.* 2004, Packer *et al.* 2009, 2011). Therefore, although consumptive use can potentially benefit human communities and conservation in NNR, the sport hunting of leopard to generate revenues should be researched further and closely monitored in NNR and elsewhere.

Despite the potential for coexistence between humans and leopard, in some areas of NNR leopard habitats will inevitably be lost in favor of more profitable land uses for communities, reducing the leopard population further. A decision to devote areas exclusively to long-term leopard conservation is critical to minimize potential conflicts with other land uses (Munthali 2007). For example, in the Mavago district where there are the most fertile soils of NNR (Cunliffe *et al.* 2009), tobacco generates larger gross revenue from direct employment and benefits more families ($529,226 for

1894 families, National Directorate of Land and Forestry (DNTF), unpublished data) than sport hunting ($130,763 for about 400 families, Chapter 2). Therefore it is unlikely that people there will be convinced to stop converting carnivore habitats in order to benefit leopard conservation. By contrast, in eastern of NNR soils are less productive, and it is there that PEC might be more successfully implemented. Considering a zoning system, such as that in Coutada 9 in the center of Mozambique (Nhacainga Conservancy Project 2011) might benefit leopard conservation and their prey in NNR. Based on this system, there would be areas exclusively used for wildlife where the tourism revenues are allocated mainly to the reserve management, a multiple use zone with some settlement where wildlife is the primary land use and where revenues from community based tourism go to villagers. The zoning can potentially also contribute to control ongoing dispersion of settlement in NNR, and prevent proliferation of farming along rivers and other sensitive habitats. Furthermore, such an approach would mean that some areas are less exposed to illegal hunting, and would provide a good framework for engagement and involvement of communities.

The zoning could also be extended to hunting concessions, thereby improving the protection of leopard populations. Building on Balme *et al.* (2010) protocol, no hunting zones inside each hunting block in NNR, large enough to accommodate viable populations of leopard and other large predators, and devoted exclusively for photographic tourism could be created. These zones would act sources of leopard for areas where leopard and other carnivores could be hunted (Woodroffe & Ginsberg 1998, Balme *et al.* 2010). Furthermore, the rotation of hunting locations inside concessions would spread the hunting pressure more evenly across the blocks (Balme *et al.* 2010) and would limit the effects of infanticide.

The long-term conservation of wide-ranging carnivore populations requires protection beyond the protected areas boundaries (Forbes & Theberge 1996). Although hunting areas can be a source of mortality for large carnivores (Loveridge *et al.* 2007), if well managed, hunting blocks surrounding NNR can prevent additional human settlement and habitat fragmentation on NNR's borders (Lindsey *et al.* 2007a, Soto 2009). In addition, there would be an increase, by at least 23%, of the total area under protection for the conservation of carnivores and their prey in NNR.

4.1. Conclusion

The data presented in this study on the economic value of leopard at a global and local scale, level of depredation on livestock, and leopard population densities, are useful as a baseline to develop sustainable management systems for leopard sport hunting in NNR. Adverse impacts from growing human populations on leopard populations can be mitigated substantially by integrating and refining the zoning principle presented in this thesis and improving revenue distribution to local partners. My findings have consequences beyond leopard conservation, and might be useful for future land-use development policies and approaches in northern Mozambique. Furthermore, measures proposed in this study to protect and conserve leopard in NNR might have impacts beyond the reserve, by creating one of the largest and contiguous areas under protection available for conservation of leopard and other large carnivores in Mozambique and in Africa.

4.2. Recommendations

- While I determined the proportion of revenues accruing to NNR communities, future studies should investigate more specifically how the revenues contribute to improve the livelihoods of rural villagers and conservation management in concessions (Leader-Williams 2009).
- At the community level, it will be critical to investigate mechanisms for improving alternative livelihoods linked to the sport hunting industry in NNR such as promotion of local arts, craft and pottery to be built around the hunting industry in NNR;
- The sport hunting industry is growing in Mozambique therefore it is critical to explore the viability of developing local taxidermist capacity to increase the retention of revenue in-country, and consequently increasing the economic impact of sport hunting in the national economy.
- Leopard conservation will benefit from continued monitoring of their populations, particularly close to villages and inside hunting concessions. Future studies assessing the impact of sport hunting on leopard populations in NNR should place camera traps across a range of sites with different levels of hunting. Furthermore, leopard hunting quotas may need to be adjusted locally when their numbers begins to decline;
- To reduce the pressure on leopard prey, it is critical to raise the penalties of involvement in bushmeat hunting, and to provide access to alternative protein resources to villagers dependent on bushmeat, either through domestic livestock or game farming.

- A market survey in Nampula, Niassa and Cabo Delgado provinces, where wildlife products are commercialized, would indicate how these markets influence leopard poaching in NNR.
- It is critical to work with communities to improve their attitudes towards the value of large carnivores and build tolerance through environmental education (Schumann *et al.* 2008). However, despite their poverty, communities within NNR valued leopard for economic, ecological and cultural reasons. This is in contrast with the majority of people who disliked leopard, or considered it as a pest, in South Africa and in Tanzania (Dickman 2008, Chase-Grey 2011). This suggests that there are opportunities to increase persistence of leopard in NNR because they have cultural value to the local communities (Karanth *et al.* 2010b).

Table 4.1. Potential monetary annual benefits (US$) generated to NNR local communities through Payments to Encourage Coexistence (PEC) based on the 2010 data.

Source of revenue	Value
Leopard community safaris[a]	92,000
Plain's-game safaris	2,000
Photographic and sport hunting tourists	20,000-40,000
Travel agencies and taxidermists	30,000
Others (goat sales and local employment on deep and pack of trophies)	4,000
Total	**148,000 - 168,000**

[a]revenues generated from four leopard safaris.

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APPENDICES

Appendix 2.1. Description of the data used to calculate the direct expenditure of clients charges on various components of hunting safaris in NNR during the 2010 season.

Safari component	**Data**	**Description**	**Data source**
Safari hunting package	Safari package	Cost of daily rate and trophy fee for hunting packages purchased by clients.	Operators
	Trophy royalty fee	Fees charged by the reserve management for trophy licenses purchased in MITUR.	SGDRN
	Safari hunt return form (HRF)	Information on safari types and length, species hunted, number and nationality of clients and observers involved in the safari.	SGDRN
	Licenses for clients, professional hunters, and trophy licenses	Amounts paid for client hunting licenses, government trophy licenses included in the safari package selected by clients, and license for the professional hunter guiding the safari.	Ministry of Tourism (MITUR)
Travel and accommodation costs	International airfare	Client expenditure on travel from United States of America (USA) / Europe to Niassa/Pemba (Mozambique) and return.	Travel agencies
	Accommodation	Amounts paid for nights spent in hotels in Maputo, Cabo Delgado or Niassa before and after the hunts.	Hotels
	Air charter fees	Cost of air charter from Niassa/Cabo Delgado/Lilongwe to the hunting destination in NNR.	Operators
	Licenses and visa	Client expenditure on visas and licenses for firearm and ammunition.	Ministry of Interior (MINT)
	Private custom agents	Amount charged to process client firearm documentation prior to their arrival in Mozambique.	Operators
Export costs and taxidermy	Government export licenses	Expenditure on CITES/Non CITES permits, veterinary certificates and ownership certificate for trophies hunted in NNR.	Ministry of Agriculture (MINAG)/ MITUR
	Documentation handling and packing of trophies	Amounts charged to assist client obtain all government export permits at provincial and central levels, pack of trophies and export to taxidermists.	Trophy shipping companies
	Process and mount trophies	Amounts charged to process and mount clients trophies.	Taxidermist companies

Appendix 2.2. Questionnaire used to query Niassa Reserve Hunting operators on the leopard value and economic benefits from leopard sport hunting.

A: RELATED TO CLIENTS

1. How many clients visited your block in 2010?
2. How many booked to hunt leopard?
3. How many of your clients spent a night in Mozambique before going to the block?
4. What category of hotel do your clients use before arriving in your camps?

   3 stars ☐ 4 stars ☐ 5 stars ☐ others ☐

5. How many nights did your clients that hunted leopard spend in hotels in Mozambique during the 2010 season (outside of your camps)?
6. How many non hunting guests came with your leopard hunting clients?
7. Did any of your leopard hunting clients spend time at other Mozambique resorts (fishing, diving etc)?

   yes ☐ no ☐ other ☐

B: RELATED TO SAFARI OPERATIONS

1. How much was spent on wages for staff?

   nationals ☐ foreigners ☐

2. How much was spent on social security?
3. How many workers did you employ in 2010?

Total national	☐	Total foreign	☐
Permanent	☐	Permanent	☐
Seasonal	☐	Seasonal	☐

4. What is the average cost of air charters to your camp?
5. Rank the species below (1-4) according to their importance for your business?

   lion ☐ buffalo ☐ leopard ☐ elephant ☐

6. What is the overall value of a leopard safari (total daily rate + trophy fee) in your concession?

7. Rank 1-5 what would happen financially in 2011 if you had no leopard on quota with current lion, buffalo and elephant quotas?

slight reduction in the gross income for the company	
business would not be economically viable	
there would be no change	
reduction on workforce and activities	
other:	

C: OTHER

1. Rank (1-5) how Niassa Reserve benefits from sport hunting operations in your concession.

income for management of the Reserve	
increase of anti-poaching effort in the Reserve	
employment for communities in the Reserve –incentive for conservation	
protection of a significant portion of wilderness in the Reserve	
other:(specify)	

2. Rank (1-5) the challenges to sport hunting in Niassa Reserve.

human population growth and expanding agriculture	
illegal logging	
illegal mining	
poaching of wildlife	
over regulation of activities	
high operational costs due to logistics	
theft of trophies and equipment in the camp	
too low quotas for key species:	
game populations too low	
inconsistent quota allocation by MITUR	

3. What do think that could be potential negative consequences of sport hunting in Niassa Reserve?

Note: Hunting concessionaires attributed the same rank to more than two responses available for questions B:7, C:1-2. Therefore these questions were excluded from further analysis in Chapter 2.

Appendix 2.3. Questionnaire used to query Niassa Reserve villagers on leopard poaching and conflict issues. This questionnaire was administrated in Portuguese in NNR villages and only translated into English to be included as appendix of this thesis.

Select one option for each multi choice question presented below.

I. PERSONAL DATA

1. Age:

children (7-12 years)	☐	adult (18-40 years)	☐
youth (12-18 years)	☐	old (>40 years)	☐

3. Sex:

male ☐ female ☐

4. Occupation:

farmer	☐	scout	☐
fisherman	☐	government employee	☐

II. LIVESTOCK

1. Do you have goats? If yes how many?

____

2. Where do you keep your goats during the nights?

corral ☐ yard (outside corral) ☐ field (outside corral) ☐ other_______ ☐

3. Did leopard attack your goats?

yes ☐ no ☐

4. If yes, how many goats were attacked?

____

Have you seen any snare line in the bush in 2010? If yes how many?

0-2	☐	2-4	☐	4-6	☐	6-8	☐
8-10	☐	10-15	☐	15-20	☐	20-25	☐

9. How can we prevent leopard attacks to goats in the village?

keeping goats in the corral	☐
removing leopard from the reserve	☐
using dogs to protect goats	☐
other____________________	☐
do not know	☐

III. LEOPARD

1. What is the good use of leopard in Niassa Reserve?

eat other animals	
bring tourists that come to photograph and hunt	
produce a beautiful skin that the community can use	
produce medicine for people use	
other____________________	
do not know_	

2. What type of medicine leopard produce?

luck	
alleviate headache	
give courage	
other____________________	
do not know_	

3. What are the animals eaten by leopard?

baboon		kudu		ant bear		reedbuck	
impala		warthog		buffalo		eland	
porcupine		waterbuck		elephant		hartebeest	
bushbuck		guinneafowl		duiker(s)		zebra	
bushpig		rabbit		francolin			

4. Why leopard attack goats in the village?

leopard are hungry and do not have other food	
there are too many goats in the village	
there are too many leopard in the reserve	
goats sleep outside corrals	
other____________________________________	
do not know	

5. How many leopard do you think are illegally hunted in the village per year?

0 - 2		2 - 4		4 – 6		6 – 8		8 – 10	
10 - 12		12 – 14		14 – 16		16 – 18			

IV. COMMUNITY PERCEPTION REGARDING LEOPARD

1. Do you like leopard?

yes		no		indifferent		do not know	

2. If you find a leopard skin in the bush and I ask to buy the skin, for how much (Meticais) would you sell it to me.

500 – 1000		1000-2000		2000-3000		3000-4000		4000-5000		5000-6000	
7000-8000		8000-9000		9000-10000		10000-15000		15000-20000		20000-25000	

V. BUSHMEAT HUNTING AND BENEFIT SHARING

1. How many times did you eat meat last week?

2. What curry do you like to feed your children with?

chicken		fish	
goat		fresh bush meat	
beans		dried bush meat	

3. If we decide to organize a party, can we buy bushmeat in the village?

____

4. What is the origin of amount of 20% distributed among communities?

government		mbatamila			concessions reserve others	

5. Does the community benefit from tourism in the Reserve?

6. What can be done for communities get more benefits from tourism activity?

Construction of public infrastructure		Give money to individuals		Increase employment		Increase number of concessionaires	

Note: NNR villagers selected more than three responses available for questions III:3, V:2, 4 and 6. Therefore these questions were excluded from further analysis in Chapter 2.