HUMAN IMPACTS ON CARNIVORE BIODIVERSITY INSIDE AND OUTSIDE PROTECTED AREAS IN TANZANIA

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DECLARATION

I, Maurus January Msuha hereby declare that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated. The material contained in this thesis has not been previously submitted for a degree at University College London or any other university.

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SUMMARY

Conservation of biodiversity throughout the world is often characterized by the establishment of protected areas. The implementation of this approach is extremely challenging particularly in developing countries due to expanding human population and demand for resources. Yet, information that is needed to guide managers and policy makers to develop effective conservation strategies is scarce in most of these countries. This thesis aimed to explore the impact of human activities on carnivore biodiversity inside and outside Tarangire National Park in Tanzania using camera traps, and to assess attitudes of agropastoralists towards carnivores using interviews. Results showed no significant difference in carnivore species richness between the park and communal grazing areas outside the park, but was a significantly different between the park and cultivated areas outside it. Non-carnivore species richness was significantly higher outside the park in grazing areas than either within the park or in cultivated areas. However, relative abundance of both carnivores and non-carnivores were both significantly higher in the park than in either grazing areas or cultivated areas outside the park. These variations in species richness and relative abundances are apparently due to differences in the intensity and extent of human use between these areas. Estimation of species absolute abundances targeted individually identifiable species in the park only. Results showed that density of animals per 100 km^2 was: leopard (7.9 ± 2.09), serval (10.9 ± 3.17), and aardwolf (9.0 ± 2.54). No estimates were obtained for spotted hyaena and common genet due to a lack of recaptures, while variation in trail density, prey availability, and camera spacing appear to influence species capture. Attitudinal surveys revealed a low level of wildlife-related benefits and reported levels of conflict were generally high despite low levels of livestock depredation, suggesting other factors such as demand for land might be important in the reported conflict.

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LIST OF ACRONYMS

BA	Banded mongoose
BF	Buffalo
BJ	Black-backed jackal
BT	Bat-eared fox
CA	Caracal
CG	Common genet
CH	Cheetah
CMR	Capture-Mark-recapture
CV	Civet (African civet)
DW	Dwarf mongoose
EG	Egyptian mongoose
EL	Elephant
GR	Giraffe
HB	Honey badger
LG	Large spotted genet
LN	Lion
LP	Leopard
IM	Impala
PAs	Protected Areas
SE	Serval
SL	Slender mongoose
SP	Spotted hyaena
ST	Striped hyaena
UK	United Kingdom

WC	Wild cat
WCS	Wildlife Conservation Society
WD	Wild dog
WH	White-tailed mongoose
WMA	Wildlife Management Areas
WT	Warthog
ZB	Zebra
ZO	Zorilla

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1.1 Developing and Applying Camera Trapping to a Savannah Carnivore Community

Current threats to the world's biodiversity call for the need to have effective conservation strategies. However, developing effective biodiversity conservation strategies requires knowledge on species richness (the number of species) and abundance (the number of individuals of a species) in the areas that we want to conserve. This is important because species richness is frequently used in the establishment and management of protected areas (Margules and Usher 1981, Baskin 1994) and to assess whether management strategies are successful (Thomas 1996, Bawa and Menon 1997, Yoccoz et al. 2001). Information on species abundance is required for a variety of reasons e.g. assessing viability of threatened species (Linkie et al. 2006), setting of quota for species that are hunted (Baldus and Cauldwell 2005, Lindsey et al. 2007, Lindsey 2008) and monitoring populations of keystone species i.e. species that have impacts on others, often far beyond what might be expected from a consideration of their biomass or abundance (Simberloff 1998). For instance, in the Serengeti ecosystem wildebeest (*Connochaetes taurinus*) are a keystone species that determine what happens in the rest of the ecosystem and because of their importance in the ecosystem they have been monitored for a long time (Mduma et al. 1999). Information on species abundance is also essential for managing populations of species that can cause environmental damage or those which can have negative impacts on people's livelihoods (Morellet et al. 2007). For example, in South Africa, African elephants (*Loxodanta africana*) were culled in the Kruger National Park in order to control their impact on vegetation and its consequence on the depletion of biological diversity (Van Aarde et al. 1999). Similarly in the UK badgers (*Meles meles*) were culled in order to control the spread of bovine tuberculosis (*Mycobacterium bovis*) to cattle

(Tuyttens et al. 2000). Although information on species richness and abundance is important for conservation planning, our knowledge of species richness and abundance is extremely poor in many areas that are also important for biodiversity conservation (Margules and Pressey 2000). This is particularly true for carnivores, because many are nocturnal and shy, and large species tend to occur at relatively low densities, making them very difficult to study with methods such as line transects or interviews (Wilson et al. 1996, Sargeant et al. 1998, Stander 1998). In recent decades camera traps have been increasingly used to survey mammals (Rowcliffe and Carbone 2008, Tobler et al. 2008), because they can be used in a variety of environments and have worked well for many species (Champion 1992, Griffiths and van Schaik 1993, Karanth and Nichols 1998). They are also relatively cheap and more importantly photographs obtained are recognisable. Camera traps have been used successfully for making inventories of medium-sized and large mammals (Tobler et al. 2008), for estimating species relative abundances using photographic rates (Carbone et al. 2001) and for calculating occupancy i.e. the proportion of area occupied by a species (MacKenzie et al. 2002). Camera traps have also been used to estimate absolute abundance of individually recognisable species such as tiger (*Panthera tigris*) (Karanth and Nichols 1998) and puma (*Puma concolor*) (Kelly et al. 2008) using capture-mark-recapture (CMR) techniques.

However the use of camera traps as a tool for monitoring mammals to date has mainly been carried out in forest habitats e.g. in India (Karanth and Nichols 1998, Karanth et al. 2004a, Karanth et al. 2004b), Indonesia (Linkie et al. 2006, Linkie et al. 2007, Linkie et al. 2008), Belize (Dillon and Kelly 2007, Kelly et al. 2008) and in Tanzania (De Luca and Mpunga 2005, Rovero et al. 2005, Rovero et al. 2008, Pettorelli et al. submitted). Furthermore many of the previous camera trap studies have also mainly focused on single species e.g. ocelot (*Leopardus padalis*) (Dillon 2005, Dillon and Kelly 2007), leopard (*Panthera pardus*) (Henschel et al. in press), tiger (Karanth and Nichols 1998, Linkie et al. 2006) and puma (Kelly et al. 2008). The potential of camera traps to survey carnivores in areas of higher mammal biodiversity such as the African savannahs is not known. Therefore assessing the potential of camera traps as a tool for surveying mammals in the savannahs is important in order to understand its strength and weakness. Studies suggest, for example, that knowledge on species ecology is fundamental in the application of camera traps for surveying species. For instance, knowledge on species home range is important in determining how far apart camera traps should be placed, and body size may be used to determine optimum height for the camera above the ground. Similarly knowledge on species demographic ecology is important in determining how long camera trapping period should be (Karanth 1995, Karanth and Nichols 1998, Karanth and Nichols 2002, Silver 2004).

1.2 Carnivore Ecology

1.2.1 Diet

Carnivores are classified into two groups based on their dietary needs: (i) species which depend on meat for a high proportion of their diet; and (ii) species that feed on insects or foliage/fruits (Estes and Goddard 1967, Gittleman and Harvey 1982, Carbone et al. 1999). For example, some carnivores such as cats and weasels are strictly carnivorous while others e.g. canids and mustelids and many viverrids subsist largely on insects (Estes 1991, Carbone et al. 1999). This variation in diet may be because of difference in energy requirements between species. For instance, small carnivores usually have lower energy requirements than large carnivores and hence the latter require larger prey to meet higher energy demands, while for small carnivores invertebrates can provide sufficient energy (Gittleman 1985, Carbone et al. 1999). It has been shown that at around a body mass of 20-25kg, carnivores show a transition from feeding on small prey of less than half the predator's body mass to large prey that is near or above the predator's body mass (Carbone et al. 1999). Some carnivores are also more specialised in the type of prey they take – for instance, aardwolves feed strictly on two types of termites, *Trinervitermes* and *Hodortermes,* (Richardson 1987a, Kingdon 1997, Williams et al. 1997) while some species are more specialised in the size of prey they take, e.g. wild dogs show a clear selection for medium-sized antelopes (Gittleman and Harvey 1982, Gittleman 1985, Estes 1991).

1.2.2 Social Structure

Carnivores also have a variable social structure. For example, most mustelids such as the honey badger (*Mellivora capensis*) are solitary and have brief social encounters for breeding only (Begg and Begg 2005). Others, such as the spotted hyaena (*Crocuta crocuta*), live in extended social groups comprising of as many as 80 individuals (Gittleman et al. 2001). Social carnivores benefit from the complex behavioural organisation of such groups, which may involve hunting together, taking care of each others young and/or protecting the territory held by the group (McCloud 1997). For instance, in the Serengeti it was shown that adolescent cheetahs living in temporary sibling groups had higher survival than singletons, while adult male cheetahs living in coalitions had higher survival than singletons in periods when other coalitions were numerous (Durant et al. 2004). It has also been found that group living in social animals such as lions provides stability to the population under changing environmental conditions such as prey availability and vegetation cover (Packer et al. 2005).

1.2.3 Demography

Carnivore reproductive ecology is extremely varied. Many species, particularly the large ones, have low reproductive rates (Sillero-Zubiri and Laurenson 2001). For instance, the average litter size for the cheetah in the Serengeti is 3.6 (Laurenson 1994) and cheetah litters rarely exceed six in the wild (Caro 1994). Female cheetahs start to breed in the wild when they are about two years old (Kelly and Durant 2000), and the males become sexually mature at almost the same age (20-23 months) (Caro and Collins 1987). Similarly, some black bears produce only one offspring every seven years, which is in marked comparison to some mongooses, which can produce three litters or eight young in a single year (Gittleman et al. 2001). However it is important to note that many reproductive strategies in mammals, such as timing of mating periods, fertility rates and litter sizes are largely influenced by environmental conditions (May and Rubesnstein 1987). For example, food availability may affect body conditions of females and thus affect the age at which they start breeding, as well as the resultant litter size (Langvatn et al. 1996, Hardy 1997). Nonetheless, many carnivores are long-lived e.g. cheetahs in Namibia have shown to live up to twelve years in the wild (Marker et al. 2003a).

1.2.4 Competition among Carnivores

Many carnivores compete with each other and competitively inferior species may seek to escape this competition by using refuge areas or habitats which do not overlap with home ranges of their competitors (Shorrocks 1991, Palomares and Caro 1999, Caro and Stoner 2003). For example, cheetahs have low competitive ability compared to their principal competitors, spotted hyaenas and lions (*Panthera leo*) (Hofer and East 1995, Durant 2000). All three predators partly rely upon migratory prey species (Hofer and East 1995), and because of the patchy distribution of lions and hyenas, cheetahs persist in the ecosystem by employing predator avoidance behaviour. However, in order for the avoidance to be successful the presence of heterogeneous habitats is important (Durant 1998). Wild dogs also show similar behaviour in the presence of large carnivores in areas that are heavily used by lions (Creel and Creel 2002). Predator avoidance thus plays an important role in structuring species communities by promoting coexistence (Lima and Dill 1990, Durant 2000), as do strategies such as variation in dietary requirements (Gittleman and Harvey 1982; Estes 1991; Carbone et al. 1999), and the use of heterogeneous habitats (Durant 1998). Studies suggest that mammalian top predators are key determinants of trophic structure and biodiversity in many terrestrial ecosystems (Palomares and Caro 1999, Terborgh et al. 1999, Sinclair et al. 2003, Hebblewhite et al. 2005). This is because top predators have an impact on herbivore communities and on predators in lower trophic levels (Palomares and Caro 1999). It is argued that a reduction in the abundance of large predators can lead to an increase in diversity and population of medium-sized carnivores in an area (Groom et al. 2006). This increase in the abundance and diversity of mesopredators can lead to an increase in the predation of smaller and more vulnerable species, which may lead to extinction and therefore loss of biodiversity (Crooks and Soule 1999, Sanicola 2007). However, there are only a few studies which have actually demonstrated this phenomenon (Elmhagen and Rushton 2007), probably because carnivores are difficult to study.

1.3 Carnivore Biodiversity

The order Carnivora contains 271 species (Wilson and Reeder 1993), 70 of which are found in Africa (Mills et al. 2001). Carnivores are unusual compared to other taxa such as primates because they are found almost in every type of habitat (Gittleman et al. 2001). Despite this, species richness i.e. the number of species, is generally lower in the order Carnivora compared to other taxa – for instance, there are 394 species of primates (Mittermeier et al. 2007) compared to 271 species in the order Carnivora. The relatively low carnivore species richness is due to carnivores occupying higher trophic position than their non-carnivore prey. Species richness tends to be higher at lower tropic levels (Petchey et al. 2004, Raffaelli 2004) because at higher trophic levels species tend to occur at relatively small population sizes and are more vulnerable to extinction due to demographic stochasticity and environmental changes (Lande 1993).

1.4 Impacts of Human Development on Carnivores and Other Biodiversity

The interactions between people and wildlife play a pivotal role in shaping the perceptions of people and the development of conservation strategies. Therefore understanding the nature of these interactions is central to the development of effective conservation plans and may be beneficial to both humans and wildlife (Happold 1995). For example, where large carnivores are visible, they can attract visitors and hence provide an important source of foreign revenue, especially for developing countries (Treves and Karanth 2003). On the other hand, large carnivores can cause bodily harm to humans, prey on livestock and can act as reservoirs of diseases which affect humans and their domestic animals, particularly dogs (Happold 1995, Cleavaland et al. 2001). Humans also affect large carnivores through land conversion for agriculture and human settlements (Bauer and Van der Merwe 2004), through hunting of species for either subsistence, sport or trophies and through depletion of prey species (Caro et al. 1998, Baldus and Cauldwell 2005, Lindsey et al. 2007).

1.4.1 Land Use and Climate Change

It is widely accepted that global biodiversity is changing at an alarming rate (Millennium Ecosystem Assessment 2005), and that much of this change in biodiversity is induced by human activities (Pimm et al. 1995). Of all human impacts on biodiversity, land use change has been singled out as the greatest immediate threat to terrestrial biodiversity because it results in fragmentation and loss of habitats (Vitousek et al. 1997, Sala et al. 2000, Jetz et al. 2007). Such changes may lead to restriction of animal movements as well as decline in species richness and abundance. Existing evidence shows that land use change has negative impact on species e.g. predictions of the impact of tropical forest clearance show that by about 50,000 species may become extinct by 2060 (Pimm and Raven 2000). Similarly the 'human footprint' study (Sanderson et al. 2002) suggests that anthropogenic land transformation is the single greatest threat to biodiversity. Furthermore it is also estimated that 86% of globally threatened mammals on Earth are at risk from habitat change (Baillie et al. 2004).

There are many anthropogenic factors that drive land use change. The most important ones include the need for human settlements, cultivation of crops and other economic activities (Geist and Lambin 2002). The impacts of these drivers of land use change on biodiversity are different because they differ in the extent to which they modify the quality of habitats (Goudie 1986, Forman 1995). However, land use change due to agricultural expansion is often cited as one of the major threats to biodiversity. The Millennium Ecosystem Assessment (2005) report draws particular attention to the expansion of crop land across the globe and points out that more land has been converted to agriculture after 1950 than the years before. Generally it is predicted that the impact of land use change on biodiversity will have a much greater effect on tropical countries. This is because species in the tropics tend to have smaller home ranges than those at higher latitudes due to higher diversity of habitats in the tropics (Jetz et al. 2007). It is also possible that human population growth and land use change are progressing faster in tropical countries (Ceballos and Ehrlich 2006), and land use plans are lacking in many areas.

In addition, predictions also suggest that the impact of land use change on biodiversity will be even more severe in the future because land use change affects land cover which ultimately affects climate (Jetz et al. 2007), and climate change affects precipitation patterns and hence overall primary productivity of ecosystems and species richness (Higgins 2007, Thuiller 2007). Large carnivores are particularly vulnerable to habitat loss because they have large home ranges and require extensive, intact habitats to survive (Sillero-Zubiri and Laurenson 2001). For example, the loss of habitat is cited as the main threat to cheetahs (Caro 1994), partly because cheetahs are more vulnerable to spatial fragmentation since heterogeneity in habitat is required for successful predator avoidance (Durant 1998). Furthermore, habitat loss may affect carnivores indirectly by reducing the availability of prey. Carbone and Gittleman (2002) showed that the abundance and distribution of carnivores is strongly related to the population density of their prey species. However, the impact of loss of habitat may be more severe for some species than others, yet to date there are no comprehensive studies that have investigated the impact of habitat loss on carnivore biodiversity, especially in areas which have rich carnivore community such as Tanzania. Understanding how land use change affects species is an important first step towards developing conservation plans.

1.4.2 Hunting

Hunting is another way through which humans have impact on wildlife. Although the practice has existed for millenia, hunting, whether subsistence or commercial/trophy hunting, can have significant impact on wildlife populations e.g. the wild boar (*Sus scrofa*) was hunted to extinction in some parts of Europe such as the British Isles and Scandinavia (Apollonio et al. 1988, Oliver et al. 1993). Hunting can also lead to changes in species' community composition if individiuals of specific age groups or sex are removed. This can reduce birth synchrony, delay development of body mass, alter offspring sex ratio or destabilise social structure (Milner et al. 2007). However, the impact of hunting on mammal populations varies depending on the species and hunting method used (Fa et al. 2005), as well as the productivity of the area. For instance, tropical forest ecosystems are less productive than savannahs and therefore are more vulnerable to uncontrolled exploitation (Fa and Peres 2001). Some species are also more affected by hunting than others. For instance, in Tanzania, a study showed that density species such as eland (*Taurotrogus oryx*), kudu (*Tragelaphus sp*), reedbuck (*Redunca arindinum*) and lion and leopard (*Panthera pardus*) were impacted more than other species due to tourist hunting (Caro et al. 1998). This is probably because these species have higher economic value for trophies and overexploitation of such species is likely to occur (Child 2000). However it is also true that overexploitation can also be due to illegal hunting as most illegal hunters target large mammals in order to obtain large quantities of meat (Fischer and Linsenmair 2001, Naughton-Treves et al. 2003b).

1.5 Impacts of Carnivores on Human Development

Carnivores come into conflict with humans for a wide variety of reasons. First and foremost, people often see large carnivores as a threat to human life because of real and perceived predation on humans. Second, carnivores prey on livestock causing considerable economic losses to humans. Third, carnivores also prey on game which humans eat and therefore compete with humans (Inskip and Zimmerman 2009). Identifying the sources of these conflicts and assessing the attitudes of humans towards carnivores is fundamental for developing effective conservation strategies, as negative attitudes are a major driver of carnivore persecution throughout the world (Woodroffe and Ginsberg 1998, Hussain 2003, Woodroffe and Frank 2005).

1.5.1 Livestock Depredation

Human-carnivore conflict over livestock depredation is a serious management issue that wildlife managers are facing today (Ogada 2003, Patterson et al. 2004, Graham et al. 2005, Zimmerman et al. 2005). For example, it is estimated that over 75% of the world's felid species are affected by conflict with people. The severity of the conflict has also been found to increase with species body mass and points out in particular nine species as being most important for conflict with people. These are: caracal (*Felis caracal*), cheetah (*Acinonyx jubatus*), Eurasian lynx (*Lynx lynx*), jaguar (*Panthera onca*), leopard (*Panthera pardus*), lion (*Panthera leo*), puma, snow leopard (*Uncia uncia*) and tiger (Inskip and Zimmerman 2009). However it is also important to note that some other species may be locally important as a source of human-wildlife conflict and they may not feature at a global scale. For example, angry farmers in Norway were reported to have killed wolves to reduce sheep depredation (Røskaft et al. 2003), but at present the wolf is not seen as a species that has significant impact on livestock depredation at a global scale (Inskip and Zimmerman 2009). In Africa, killing of carnivores because of livestock loss has been widely reported e.g. between 1980 and 1990 at least 320 lions were killed on farms bordering Etosha National Park in Namibia (Berry 1990), leopard killing by farmers due to livestock depredation has been reported in the Cape Province in South Africa (Stuart et al. 1985) and in Kenya at least 14 spotted hyenas were reportedly poisoned in a single incident in the Maasai Mara National Reserves, apparently in an attempt to reduce livestock depredation (Holekamp and Smale 1992). Losses due to depredation are more common with cattle, sheep and goats (Inskip and Zimmerman 2009). Such losses can be very severe and may significantly affect local people's livelihoods and therefore their support for conservation. The scale of these losses to livestock depredation is not well understood, probably because quantification of economic losses is difficult. For instance, out of 225 sources of literature reviewed by Inskip and Zimmerman (2009) only 10% contained information on economic losses from livestock depredation.

However it is also equally important to understand that sometimes carnivores are killed because of perceived conflict even if the actual levels of depredation are not high e.g. in Namibia farmers intensively remove cheetahs in order to lessen the risk of depredation, although studies show they select indigenous game (Marker 2002). Game farmers are actually more antagonistic towards cheetahs, as they view the cheetah as a major problem because of the risk of depredation on valuable introduced game species such as roan antelope (*Hippotraggus equines*) (Marker-Kraus 1997).

The reasons for carnivores preying on livestock vary between areas. In the French Jura, livestock predation by lynx was found to be strongly correlated with environmental characteristics, such as the proximity of farms to forest areas and the availability of prey, particularly roe deer (*Capreolus capreolus*). Many individuals were found to feed on roe deer despite sheep being abundant and the sheep that were attacked by lynx were those that were found very close to the forest (Stahl et al. 2002). In northern Portugal livestock depredation by wolves was shown to be linked to a scarcity of wild prey (Vos 2000). Generally it is widely acknowledged that livestock depredation often tends to be higher when wild prey availability is less abundant (Polisar et al. 2003, Bagchi and Mishra 2006, Johnson et al. 2006, Inskip and Zimmerman 2009), although in some areas, predators may learn that livestock are easier to catch, leading some individuals to switch from natural prey to hunting livestock (Mizutani 1999, Woodroffe and Frank 2005).

1.5.2 Diseases

The interactions between humans and wildlife can be affected by diseases that can infect humans and their livestock and vice versa. Diseases that originate in wildlife reservoirs can be a source of human-wildlife conflict particularly in rural communities that depend on livestock production for their livelihoods (Cleaveland et al. 2000, Cleaveland et al. 2001). For example, in East Africa pastoralists suffer serious losses as a result of disease transmission between wildlife and domestic stock, such as malignant catarrhal fever (Thompson 1997) and East Coast fever (Homewood et al. 2006). Similarly, domestic animals play a role in transmission of diseases that affect wildlife e.g. domestic dogs are a contributing factor in the transmission of canine diseases (Butler et al. 2004). As reservoirs of rabies, canine distemper and parvovirus, domestic dogs were partly responsible for the extinction of the African wild dogs in the Serengeti ecosystem (Woodroffe and Ginsberg 1999) and have also been a major factor in the population crash of critically endangered Ethiopian wolves (*Canis simensis*) (Sillero-Zubiri et al. 1996).

1.5.3 Attacks on Humans

Wildlife attacks on humans are common in some areas, although the perception of threat to humans is often greater than the real threat (Ginsberg 2001). Big cats, particularly tigers, African lions and mountain lions (*Felis concolor*) account for most of the human deaths by predators (Caro 1992, Sillero-Zubiri and Laurenson 2001). Tigers are probably the most persistent problem species in this regard and pose a continuing problem in some parts of India where people and tigers use the same habitat (Seidensticker and Lumpkin 1992). Encroachment of humans into areas which were predominantly used by wildlife has been highlighted as the underlying reason for increasing attacks by big cats (Foreman 1992, Seidensticker and Lumpkin 1992, Packer et al. 2005) as well as the depletion of wild prey due to human encroachment (Reza et al. 2002, Packer et al. 2005). In Tanzania it was shown that lion attacks on humans increased due to reduction of natural habitats and depletion of prey caused by human encroachment (Packer et al. 2005). However it has also been reported that old lions or those with dental problems such as tooth breakage are more likely to attack humans, because they are incapable of normal predatory behaviour (Patterson et al. 2003, Baldus 2006). Wildlife attacks on people, even if rare, can have significant effects on conservation programmes that require the support of local communities, as they clearly elicit serious conflict.

1.6 Global Carnivore Conservation Status

It is clear from above that the conservation of biodiversity throughout the world is extremely challenging due to expanding human populations and the associated impacts on wildlife. These challenges are particularly acute in sub-Saharan African countries which are currently characterised by a rapid increase in human populations (Ceballos and Ehrlich 2006), and unfortunately it is also where information for conservation planning is scarce for most species (Rodriguez and Delibes 2003). These challenges are even bigger for carnivores because the populations of many species are declining very fast due to loss of habitat, hunting, depletion of prey, diseases and trade in body parts as well as conflict with humans (Novaro et al. 2000, Sillero-Zubiri and Laurenson 2001). These declines are also accelerated by inherent biological factors that make carnivores more vulnerable to environmental change (Cardillo et al. 2004, Cardillo et al. 2005). For instance, large carnivores are usually at the top of food chain, which means that they will always be less abundant than their herbivore prey and therefore be more vulnerable to extinction (Noss et al. 1996, Sillero-Zubiri and Laurenson 2001). Furthermore, because of their large body size and high trophic position, large carnivores require extensive home ranges and large prey populations to survive and therefore only large and relatively intact ecosystems can support viable populations. Such intact ecosystems are difficult to maintain because of increasing human population and the associated demand for land and other resources. Consequently large carnivores tend to suffer first when human population expand into untouched habitats (Woodroffe and Ginsberg 1998, Woodroffe 2000a, Sillero-Zubiri and Laurenson 2001). In places where large carnivores still occur outside protected areas, they are often intentionally or accidentally killed by humans, which can limit their persistence (Woodroffe and Ginsberg 1998, Graham et al. 2005, Woodroffe and Frank 2005). However this is probably more important in Africa where large carnivores are more abundant and where management may be ineffective because of a lack of sufficient financial and human capacity than in developed countries. It has been shown, for example, that large carnivore populations in North America increased after the introduction of favourable legislation despite increase in human population density (Linnell et al. 2001). Therefore given effective management structures, large carnivores can coexist in human dominated landscapes (Linnell et al. 2001).

In addition to the direct impacts of people on carnivores as discussed above, loss of habitat has been shown to have significant impact on the abundance and distribution of many species. For example, the decline of the African lion (*Panthera leo*) in central and western Africa (Bauer and Van der Merwe 2004) and African wild dogs (*Lycaon pictus*) across their entire range in Africa (Woodroffe et al. 2004) are both primarily due to loss of habitat. The loss of habitat not only affects available habitat for carnivores but also affects the availability of prey species, which in turn affects the abundance and distribution of carnivores (Carbone and Gittleman 2002). Because of these anthropogenic pressures, the conservation of carnivores to date has focused mainly on the protected area network where human densities are low (Linnell et al. 2001, Woodroffe 2001).

1.7 The Importance of Conserving Carnivore Biodiversity

The conservation of carnivores can be important to people in a variety of ways e.g. they are an important component of many ecological systems and play a role in maintaining ecosystem health (Terborgh et al. 1999, Terborgh et al. 2002, Ray 2005). Being at the top of the food chain, the presence of carnivores in an area has many important ecological consequences e.g. the regulation of prey numbers (Terborgh et al. 1999). Carnivores are also an integral part of the overall biodiversity and their relationship with prey and each other plays an important role in influencing population dynamics, behaviour and evolutionary processes in ecosystems (Mills

2005). Therefore the removal of top predators from ecosystems commonly results in dramatic changes in biodiversity and community structure, and as a result can have severe consequences for the ecosystem concerned (McShea et al. 1997, Terborgh et al. 1999). Without carnivores, the fluctuation in densities of prey species is likely to be more drastic; ungulate numbers can rise substantially during favourable times and then experience large declines during poor times, which can result in changes in vegetation and community structure (Mills 2005) and density-dependency effects (Skogland 1985). For example, in the absence of large carnivores more sensitive competitor species such as roan antelope may decline because of lack of predation on more abundant species such as buffaloes (*Syncerus caffer*), zebras (*Equus sp*) and wildebeests which exert more pressure on vegetation (Mills 2005). However, the impact of top predators on prey community is not always direct. Indirect effects also play an important role in shaping ecosystem structure (Roemer et al. 2009). For example, fear of predation can affect prey species' activity patterns, habitat use, group size, and response to predators (Lima and Dill 1990, Altendof et al. 2001). These indirect effects of predation can cause prey to forego food for their safety as they shift activities toward safer areas with less food, or they may increase vigilance at the expense of feeding efficiency. These alterations can ultimately affect the prey community (Lima and Dill 1990, Altendof et al. 2001).

Large carnivores are an important tool for conservation planning because they are often used as indicators, umbrellas, flagships or keystone species (Ray 2005). An indicator species refers to an organism whose characteristics (presence or absence, population density, dispersion, reproductive success) are used as an index of attributes that are too difficult, inconvenient or expensive to measure for other species or environmental conditions of interest (Landres et al. 1988, Simberloff 1998). Generally indicators can be sub-divided into three groups: (a) Health indicators - those indicators that can be used to assess changes in habitat health, (b) Population indicators – those that can be used assess changes in populations of other species; and (c) biodiversity indicators – those that can be used to assess changes in biodiversity (Caro and O'Doherty 1999). Umbrellas are those species that need large tracts of habitat, therefore by conserving such species many other species are automatically saved (Simberloff 1998). Large mammals such as rhinos or cougars are often used as umbrella species (Beier 1993). Flagship species are normally charismatic large vertebrate species that can be used to engage public interest and sympathy (Simberloff 1998) e.g. WWF uses the Giant Panda (*Airulopoda melanoleuca*) as a flagship species to promote conservation (Maes 2004).

It is argued that because large carnivores require extensive and intact habitats to survive then by conserving them, other species found within their range or habitats are also conserved (Ray 2005). The use of such biodiversity indicators to facilitate conservation planning is generally common (Pearson 1994, Faith and Walker 1996, Andelman and Fagan 2000), but it is important to point out that the use of biodiversity indicators has often been a subject of debate (Cabeza et al. 2008, Roth and Weber 2008, Sergio et al. 2008). This is so because there is lack of clear scientific backing to support the hypothesis that areas where top predators occur are also important for biodiversity (Sergio et al. 2006). For example, the use of raptor species to predict plant, butterfly and bird species richness in Switzerland showed that raptors were only useful in predicting plant and bird species richness but not butterflies. Tits (*Parus spp*) which are at lower trophic level performed equally to raptors in predicting species richness and were a better predictor for high butterfly species richness (Roth and Weber 2008). However at the moment studies on the use of top predators as surrogates of biodiversity are limited and therefore it may be too early to discourage the use of top predators for conservation planning (Sergio et al. 2008). Top predators can be use can be used for many other purposes in conservation e.g. some species may be selected as flagships to raise public awareness or raise funds in order to increase the number of reserves, promote connectivity corridors or enlarge existing reserves and thus conserve more species (Sergio et al. 2008).

In addition to conservation planning top predators are also charismatic and can have economic benefits, particularly in developing countries either through revenues generated by hunting (Child 2000, Baldus and Cauldwell 2005, Lindsey et al. 2007) or where they are sufficiently conspicuous, through photo tourism (Treves and Karanth 2003). Carnivores also often have socio-cultural values e.g. in some societies their products such as skin, claws and teeth are used in traditional medicine (Ntiamoa-Baidu 1992, Rowe-Rowe 1992, Ntiamoa-Baidu 1997, Begg and Begg 2005).

1.8 Protected Areas and their Importance for Conservation

Traditionally conservation of biodiversity throughout the world has been characterised by the setting up of protected areas (PAs) (Pressey 1996, Chape et al. 2005). These are areas set aside principally for the protection and maintenance of biological diversity and their natural and associated cultural resources. PAs are managed through national legal systems or in some cases through other effective frameworks (IUCN 1994). IUCN (World Conservation Union) has classified protected areas into six categories. These are: Ia: Strict Nature Reserves – managed mainly for science, Ib: Wilderness Area – managed for wilderness protection, II: National Park - managed mainly for ecosystem protection and recreation, III: Natural Monument – managed mainly for conservation of specific features, IV: Habitat/Species Management Area – managed mainly for conservation through specific intervention, V: Protected Landscape/Seascape – managed mainly for landscape/seascape protection and recreation; and VI: Managed Resource Protected Area – managed mainly for sustainable use of natural ecosystems. Generally categories I-III consist of areas where human intervention is restricted (core-protected areas), Category IV allows human intervention although the main goal remains conservation, and categories V and VI attempt to minimise human influence on land or seascapes (semi protected areas and buffer zones) (IUCN 1994).

Most PAs have strict rules that exclude human activities and this enables them to provide better protection for many species that would otherwise be difficult outside reserves due to human activities (Salafsky and Wollenberg 2000). PAs are therefore well recognised as important ‘core’ units for *in situ* conservation (Brandon 1998, Bruner et al. 2001, Balmford et al. 2002, Chape et al. 2005, Gorenflo and Brandon 2006). Nevertheless, PAs alone cannot provide a long term solution for conservation of some species, such as large carnivores, because many are too small to maintain viable populations. This is because large carnivores usually have large home ranges and therefore only large PAs can provide protection (Terborgh 1999). In central and southern America, jaguar require an area of up to 750,000 hectares for a viable population (Terborgh 1999). Yet very few of the world’s ~100,000 PAs have an area larger than 750,000 hectares (Chape et al. 2005). Moreover, many wildlife species disperse outside PAs at certain times of the year and come into contact with humans (Western and Gichohi 1993, Kahurananga and Silkiluwasha 1997, Thirgood et al. 2004), again making their survival difficult due to human activities. Furthermore, effective management of PAs requires sufficient human and financial resources and law enforcement which are lacking in many developing countries (Salafsky and Wollenberg 2000).

1.9 Protected Area Systems and Management in Tanzania

Although IUCN has developed guidelines for management of protected areas, individual countries also have their own systems of protected area classification and management (IUCN 1994). Tanzania, for example has set aside an extensive system of protected areas, covering around 20% of the total land area, with different levels of use and management structures (URT 1998b, Leader-Williams 2000).

These are:

i. National Parks - areas of high biodiversity values representing unique habitats of the country. They are managed primarily for conservation of the representative habitats and wild animals that constitute naturally occurring biodiversity in Tanzania. No consumptive use of biodiversity is permitted in these areas. National Parks cover about 4% of the total protected land surface. They are managed by Tanzania Parks, a parastatal institution in the Ministry of Natural Resources and Tourism (MNRT).

ii. Game Reserves - areas where activities such as tourist hunting, game viewing and traditional use are allowed. Others such as education and research are also permitted. No human settlement is allowed in Game Reserves. Overall these areas cover about 15% of the protected area network in Tanzania. Game Reserves are managed by the Wildlife Division of the MNRT.

iii. Ngorongoro Conservation Area (NCA): This is a unique area designated for the conservation of archaeology, culture, and wildlife and water catchment. It is also a multiple land use area, which has accommodated pastoralists and wildlife. NCA covers 1% of the protected area network. Wildlife viewing and cropping in some areas are permitted. NCA is managed by the Ngorongoro Conservation Area Authority, a parastatal institution in the MNRT

iv. Game Controlled Areas - are not strictly protected areas. They are managed for tourist hunting, resident hunting, game viewing, game cropping, and live capture. These areas allows human settlement and other human activities including research and education

(WCA 1974). Game Controlled Areas cover about 8% of the protected area network and are managed by local administrative regions.

v. Forest Reserves - areas managed for the protection of forests. Non-consumptive utilisation such as scientific research and tourism are allowed but not human settlement. They are managed by the central government through the Forest Division in the MNRT. Forest Reserves cover 15% of Tanzania's land surface of which 3% overlap with wildlife protected areas.

vi. Open Areas – these have no formal protection status but are managed for tourist and resident hunting, live capture, cropping and crop protection. Their management falls under the local administrative regions.

vii. Wildlife Management Areas – these are not wildlife protected areas per see, they are areas designated for local communities to manage and benefit from wildlife on their own lands. At the moment this scheme is in its pilot stage.

1.10 The Importance of Conserving Biodiversity outside Protected Areas

Because of the many limitations faced by PAs discussed above, long term conservation of wildlife, especially for wide ranging species, ultimately depends on the presence of intact habitats outside PAs (Woodroffe and Ginsberg 1998, Ottichilo et al. 2001, Gereta et al. 2004, Thirgood et al. 2004) as well as the support of local people who live in these areas. In East Africa, for example, it is estimated that about 70 percent of wildlife populations are dispersed outside PAs on land which are also used for pastoralism (Homewood and Rodgers 1991, Western and Gichohi 1993). Therefore the presence of unfenced and uncultivated rangelands adjacent to PAs is critically important because it maintains ecological connectivity and enhances long-term survival of wildlife (Western and Ssemakula 1981, Bennet 1998). In some cases areas outside PAs can hold important wildlife populations either seasonally or throughout the year compared to PAs. For instance, Caro (2001) has shown that a greater diversity and abundance of small mammals exists outside Katavi National Park in Tanzania than inside the park. Similarly, it has been shown that adjacent pastoral areas outside Mburo National Park in Uganda support higher densities of ungulates than the park (Rannestad et al. 2006). Moreover, some species, especially medium-sized carnivores, have been shown to be more abundant outside PAs where they can avoid high densities of large predators (Creel and Marusha 1996) e.g. wild dogs appear to be more successful outside core PAs (Woodroffe et al. 2004)

Unfortunately many areas outside PAs are threatened by increasing human pressure, especially land use change. For example, research conducted in the Maasai-Mara National Reserve in Kenya and the Serengeti National Park in Tanzania has shown that wildlife populations in Kenya have declined significantly over the last few decades, primarily due to subsistence and commercial expansion of agriculture (Homewood et al. 2001, Ottichilo et al. 2001, Serneels and Lambin 2001). The presence of buffer zones in the Tanzania side of the ecosystem helped to maintain wildlife populations (Homewood et al. 2001).

1.11 Community-Based Conservation

Because of the many problems that face biodiversity conservation today are related to human activities, thus finding a solution to these problems must also involve people. Community-based conservation (CBC) has been suggested as means for addressing these problems and consequently there are many programs that have been initiated (Western 1989, Child and Child 1991, Halladay 1995). The development of the CBC is based on the idea that development and conservation can be simultaneously achieved (Berkes 2004). For instance, the establishment of PAs sometimes, especially in developing countries involves the exclusion of people from wildlife areas and thus affect livelihoods of people who were dependent on natural resources from these areas (Salafsky and Wollenberg 2000). But it also true that sometimes the establishment of conservation areas does not necessarily impact negatively on local people. In some countries in Africa and Latin America it has been found that human population densities around protected areas have been increasing because of economic opportunities due to donor investment in conservation. These findings highlight the importance of protected areas to local people (Wittemyer et al. 2008).

Although CBC has been advocated widely as means for achieving effective biodiversity conservation, its success has been very variable. In some areas its introduction has won the support of local communities in conservation. For example, in Ostinal (Costa Rica) local communities were shown to be very supportive of conservation of sea turtles because of the economic incentives accrued from harvesting sea turtle eggs (Campbell et al. 2007). In Namibia it has been shown that CBC helps to protect wildlife and the nomadic lifestyle of the Himba people, and communities participate in activities such as antipoaching, resulting in an increase in wildlife populations in the area (Cramer 2000). In Tanzania CBC has only recently been introduced over the last two decades during which there has also been a change in policies relevant to biodiversity conservation in the country, which explicitly state the need for local community involvement in biodiversity conservation e.g. both the New Wildlife Policy (URT 1998b) and Forest Policy (URT 1998a) stress the need for community-based conservation. For example, 1998 Wildlife Policy stipulates the importance of establishment of Wildlife Management Areas (WMAs) outside protected areas for local communities to manage and receive benefits from wildlife in their lands (URT 1998b, Nelson et al. 2007). The establishment of WMAs would therefore involve demarcation of village lands and managing wildlife from which villages would receive tangible benefits (URT 1998b). To date the development and implementation of WMAs has been slow, but some are now in place e.g. the

Pagawa-Idodi WMA (PI-WMA) in the Rungwa-Ruaha landscape and Iparakuyo (JUKUMU) WMA in the Selous Game Reserve (Nelson et al. 2009) and the Burunge WMA west of the Tarangire National Park. However there have been several problems with the WMA approach. In some areas communities have invested money in protecting wildlife and setting aside land for managing wildlife and they are yet to make money in return for conserving wildlife. Recent study highlights four main problems on the implementation of WMAs in Tanzania. First, the WMA has mainly focused on wildlife rather than all natural resources such that it is difficult for example, to combine wildlife conservation and forestry. Second, some communities have refused to participate by arguing that WMAs are just a strategy for taking away their traditional lands. Third, the process is also being criticized for being bureaucratic, but also because the government has more control e.g. allocation of hunting blocks and investments in the WMAs needs the approval of the government (Nelson et al. 2009). Fourth, there is also confusion as to who manages WMAs. At the moment the structure of WMAs does not involve local governments, but instead creates institutions to manage resources which take time for such institutions to evolve and attain capacity. Generally the introduction of WMAs has received mixed feelings among local communities. In the Loliondo area Maasai pastoralists rejected a government proposal to establish a WMA because the Maasai viewed it as a way of loosing their customary land rights to conservation (Nelson et al. 2009). A similar thing happened in the Simanjiro area – Emboreet and Loibosiret villages (Sachedina and Trench 2009). In some areas e.g. around Lake Burunge the establishment of WMAs has involved eviction of people which is likely to affect people's support to conservation (Igoe and Croucher 2007). Given that local community involvement is important for achieving effective conservation, understanding what drives their attitude to conservation is extremely important because attitudes influence people's behaviour and therefore support for conservation (Fishbein and Abjze 1975, Browne-Nuñez and Jonker 2008).

1.12 Carnivore Biodiversity Conservation in Tanzania

Tanzania is a biodiversity hotspot for carnivore conservation in Africa. The country holds at least half of the species found in the continent including some of the most important populations of globally threatened carnivores such as the cheetah, the African wild dog, lion, spotted hyaena, striped hyaena (*Hyaena hyaena*) and spotted-necked otter (*Lutra maculicollis*) (Mills et al. 2001). Wildlife is an important source of foreign revenue for Tanzania and contributes about 20% of the country's GDP (BoT 2007). Despite Tanzania having a large network of protected areas, human activities still pose a major threat in many wildlife areas. Tanzania's population has risen from 23.1 million in 1988 to 34.6 million in 2002 (URT 2002) which increases demand for land, particularly for agriculture and other resources. Many of the PAs in Tanzania were established in semi-arid zones which for centuries were areas dominated by pastoralists. Traditionally pastoralists have co-existed with wildlife throughout Africa (Homewood and Rodgers 1991). However, in recent decades this co-existence has been declining due to the replacement of pastoral systems with susbsistence and commerical agriculture. The replacement of pastoral systems in Tanzania is being excerbated further by recent economic liberalisation which promotes large-scale farming (Leader-Williams 1995, Mertens et al. 2000, Gros 2002). For example, in the Tarangire ecosystem large areas of pastoral lands which are important for wildlife and livestock have been set aside for the establishment of large scale state and private farms (TCP 1997). The impact of loss of wildlife habitats has already been shown to aggravate conflict between humans and wildlife in some areas, especially lions (Packer et al. 2005). In recent years lion attacks have become more common in southern Tanzania where it is estimated that more than 563 people have been killed and 308 injured between 1990 and 2004 (Packer et al. 2005). These attacks are attributed to encroachment by people into wildlife areas and the associated loss of habitat and depletion of prey species. In order to mitigate such conflicts it is important to understand how loss of habitat affects carnivores, and at the moment there is no information on this impact of habitat loss on the country's rich carnivore community.

Livestock predation is another source of conflict that affects carnivore conservation in Tanzania. A study carried out in villages outside the Serengeti National Park showed that economic losses due to livestock predation by carnivores amounted to $12,846 per year (Holmern et al. 2007). In response to these losses many livestock keepers retaliate by killing carnivores. In the Tarangire ecosystem it has been reported that nine out of eleven livestock predations by lion resulted in retaliatory killing of lions (Lichtenfeld 2005). Carnivores are commonly persecuted because of their perceived threats to livestock and humans, even if the actual impact is low. For example, Maddox (2003) showed livestock predation by large carnivores in Ngorongoro and Loliondo areas was low when compared to natural deaths e.g. diseases, yet many livestock keepers reported using some retaliatory measures, including poison which can have negative effects on non-target species.

In addition, wildlife hunting for commercial, traditional and subsistence purposes is also common in Tanzania. Carnivores are among the species that are hunted for commercial purposes, after the hunter is issued with a hunting permit by the Wildlife Division (WCA 1974, URT 1998b), but hunting for traditional use e.g. to obtain some parts of carnivores for traditional medicine (De Luca and Mpunga 2005) and for susbsistence is often carried out illegally (Barnet 2000, Loibooki et al. 2002, Holmern et al. 2007). As the human population increases adjacent to PAs, subsistence hunting is becoming increasingly important as a source of protein and cash for many local communities (Barnet 2000, Loibooki et al. 2002, Holmern et al. 2007). In a study carried out in five villages around Kilombero Valley Game Controlled

Area it was shown that wild meat was an important source of protein for most of the households surveyed (Haule et al. 2002). However, trophy hunting is the most profitable form of consumptive wildlife utilisation in Tanzania, and a lucrative growing industry (Child 2000) generating between US $ 27.6 and US $ 36.1 million annually (Baldus and Cauldwell 2005).

1. 13 Research Aims

Despite the biological and economic importance of carnivores, to date no comprehensive studies have been carried out to investigate the impact of human activities, particularly land use, on carnivore biodiversity in Tanzania. Information on the impact of human activities is required to guide conservation planning. It is clear that the suite of human-carnivore interactions that affect carnivores is very broad and therefore solutions to carnivore conservation must also be broad. In order to achieve this, my research focused on understanding the impacts of human activities on carnivores and their non-carnivore prey. My study also aimed to assess the attitudes of Maasai agropastoralists towards carnivores in the Tarangire ecosystem. Specific aims were:

1. To determine the effect of land use on carnivore and non-carnivore species richness in the Tarangire National Park, pastoral grazing areas outside the park and farmlands outside the park.
2. To determine the effect of land use on relative abundance of carnivores and non-carnivores in the Tarangire National Park, pastoral grazing areas and farmlands.
3. To determine the density of individually recognisable carnivores in the Tarangire National Park.
4. To test the use of camera traps for monitoring carnivores and their non-carnivore prey.

5. To investigate the attitudes of the Maasai towards carnivores in the Tarangire ecosystem.

1.14 Thesis Structure

My thesis has been divided into seven chapters in order to achieve the above aims.

Introduction. This section provides a literature review on camera trapping, carnivore biodiversity, and ecology. It also describes human impacts on carnivores and other biodiversity and the importance of conserving carnivores, the nature and impacts of carnivores to people and global status of carnivores. This section also described protected areas and their importance for conserving biodiversity and protected areas systems in Tanzania. It also goes discuses the importance of conservation outside protected areas and the conservation of carnivore biodiversity in Tanzania. Finally this section end up by setting research aims and provide structure of the thesis.

Study area general methods. This section describes the study area of the Tarangire ecosystem, with a focus on three land use types, namely the Tarangire National Park, pastoral grazing areas outside the park and cultivated areas outside the park. This section also provides an overview of the general methods used in this study.

Effect of land use on mammal diversity in the Tarangire ecosystem. In this section, I describe the impacts of changes in land use practices on mammal biodiversity.

Effect of land use on mammal relative abundance in the Tarangire ecosystem. Having explored the effects of land use types on species richness and diversity, in this section I explore the effects of land use type on carnivore and non-carnivore prey relative abundances.

Estimating density of individually recognisable carnivores in the Tarangire National Park. In this section I estimate the density of carnivores using capture-mark-recapture techniques and explore the potential of camera traps for monitoring species.

Attitudes of Maasai to wildlife in the Tarangire ecosystem. The conservation of carnivores ultimately depends on the support of local communities living adjacent to protected areas. In this final data section, I investigate the attitudes of the Maasai towards wildlife and towards carnivores in particular.

General discussion. This final section provide a summary of results from data chapters 3-6 and highlights the relevance of the findings for carnivore conservation both locally and globally.

CHAPTER 2: STUDY AREA AND GENERAL METHODS

2.1 Study Area

2.1.1 Historical Background

Archaeological evidence shows that most of northern Tanzania has been inhabited by the Maasai pastoralists for many years after they drove the Tatoga people out in the mid-nineteenth century, (Arhem 1985, Adams and McShane 1992). Historically Maasai have coexisted with wildlife for centuries and their livelihood was predominantly based on cattle (Homewood and Rodgers 1991, Little et al. 1999). However, the establishment of protected areas in pastoral dominated landscapes both before and after independence has reduced the amount of grazing land available for pastoralists, as well as their access to water (Shivji 1998). For instance, in the Tarangire ecosystem the establishment of the Tarangire National Park in 1970 significantly reduced pastoral range lands including access to water in the park. This loss of pastoral rangelands in the ecosystem was aggravated further by economic liberation and structural adjustment programs which promoted the introduction of large scale farming. To the east of the Tarangire National Park, in the Simanjiro District, large scale farms were established in the mid 1980s both by the state and by private individuals (Shivji 1998). Additionally pastoral land was reduced due to small scale allocations to people by village governments in areas that are also important for livestock and wildlife (Otto et al. 1998, Igoe and Brockington 1999). Increasing human populations, both from the Maasai community and in-migrants, has increased demand for land for livestock and cultivation (TCP 1997, Igoe and Brockington 1999, Sachedina and Trench 2009). The consequence of this loss of customary lands has been a decline in per capita income from livestock holdings (Mwalyosi 1992), although this decline is also attributed to lack of livestock extension services following the structural adjustment program after 1984 (Sachedina and Trench 2009). Thus, in order to meet their livelihoods most pastoralists now are engaged in crop cultivation (O'Malley 2000), although it has been reported that cultivation is not new among pastoralists in East Africa (Homewood and Rodgers 1991, Little et al. 1999). The expansion of cultivation in the Tarangire ecosystem is considered to be a principle threat to wildlife populations in the ecosystem (TCP 1997, TWCM 2000, TMCP 2002, Bolger et al. 2008) e.g. of the six migration routes that were identified in the 1960's (Lamprey 1963, Lamprey 1964), two are now completely lost and most of the remaining are also highly threatened (Bolger et al. 2008). Consequently the disruption of migratory routes due to expanding agriculture and human settlements has resulted into decline of ungulate populations. Between 1988 and 2001 the population of wildebeest, hartebeest and oryx (*Oryx beisa*) had declined in the Tarangire ecosystem by 88, 90 and 95% respectively (TAWIRI 2001). However this decline may also be due to illegal hunting and trophy hunting which occurs outside the park (Bolger et al. 2008). Recent analysis of aerial wildlife census data for eight large census zones in Tanzania also revealed that population of large herbivores in the Tarangire ecosystem has declined significantly (Stoner et al. 2007).

2.1.2 General Description of the Study Area

The research in this thesis was carried out in the Tarangire ecosystem in northern Tanzania (Figure 1), specifically in the Tarangire National Park and to the east of the park in pastoral grazing areas and farmlands. The study area is located between $3^0$52' and $4^0$24' south and $36^0$05' and $36^0$39' east. The study area is within the semi-arid ecological zone with annual rainfall ranging from 450 – 600mm (Prat et al. 1966). Short duration rains fall from October to December, and the long duration rains are from February to May, but the rains are often very erratic and variable (Kahurananga and Silkiluwasha 1997). The Tarangire ecosystem is globally important for biodiversity conservation, and is ranked second after the Serengeti-Mara ecosystem for high concentrations of migratory mammals, such as zebra and wildebeest (Reid et al. 1998, Ludwig et al. 2008). The ecosystem also contains globally threatened species such as the cheetah (TAWIRI 2007b) and the African wild dog (TAWIRI 2007d) and is globally important for its diverse and complex savannah grasslands which extend throughout the Maasai Steppe in Tanzania (Coe et al. 1999, Olson et al. 2000). Geographically, the ecosystem covers an area of about 20,000 km^2 (TCP 1997), which includes the Tarangire and Lake Manyara National Parks, Manyara Ranch, Forest Reserves such as the Marang and Nou, Mkungunero Game Reserve and the Simanjiro and Lokisale Game Controlled Areas. It is estimated that about 85% of the Tarangire ecosystem comprises of village and private lands outside core protected areas and therefore only 15% is protected (TCP 1997). For the purpose of this study, the study area was defined as the Tarangire National Park and pastoral grazing areas and farmlands outside the park to the east in Simanjiro and Monduli districts.

2.1.3 Tarangire National Park

Tarangire National Park was first established as Game Reserve in 1961 to serve as a dry season refuge for migratory mammals in the Tarangire ecosystem (Lamprey 1963, Lamprey 1964) and in 1970 the Tarangire National Park was established. The park covers an area of 2600 km^2 (TCP 1997). The vegetation in the park consists of wooded savannah which makes up the riverine area and deciduous savannah which occupies the ridges and upper slopes (Van De Vijver et al. 1999). The Tarangire National Park is a key wildlife area in the ecosystem which provides important habitats for dry season wildlife because of the presence of the Tarangire River and wetlands such as the Silale swamps (Kahurananga and Silkiluwasha 1997, Gereta et al. 2004). The park holds the largest population of savannah elephants in northern Tanzania, with an estimated 2300 elephants (Foley 2002), and also holds important populations of carnivores such as cheetah, wild dog, spotted hyaena, lion and striped hyaena, as well as herbivores such as the lesser kudu. The topography in the park is varied, with altitude ranging from 900 – 1200m. The soils consists of dark red sandy clay loam in well-drained areas, and black cotton soils in flood plains (Kahurananga and Silkiluwasha 1997).

2.1.4 Pastoral Grazing Areas

Pastoral grazing areas are important wildlife areas outside the Tarangire National Park, consisting mainly of savannah woodlands and short grass plains. The topography consists of undulating plains ranging in altitude from 1356 -1605m. The pastoral grazing areas consist of the Simanjiro and Lokisale Game Controlled Areas, which are administered by the Wildlife Division for licensed wildlife hunting and game cropping (Kahurananga and Silkiluwasha 1997). During the wet season, a considerable proportion of the park wildlife leaves the Tarangire National Park for pastoral grazing areas for about six months, especially in the short grass plains in the Simanjiro district in Emboreet, Loiborsoit and Sukuro villages (Kahurananga and Silkiluwasha 1997, Gereta et al. 2004). This migration is driven by variation in nutrient content in the soils and vegetation within the system, together with the seasonal variation in water availability. Models have shown that phosphorus levels in Tarangire are insufficient to support lactating female ungulates, forcing them to migrate to areas such as the Simanjiro plains, which contain high concentrations of phosphorus and crude protein (Voeten 1999). Therefore the existence of pastoral rangelands outside the park in Simanjiro is of great importance for the long term survival of wildlife in the ecosystem. However, this is only possible if local communities are willing to support conservation. It is therefore critical that community-based conservation should provide a means for winning support from local communities (Mehta and Kellert 1999, Infield and Namara 2001). In the Tarangire ecosystem a pilot scheme for Wildlife Management Areas (WMAs) is being established (Geofrey Mng'ong'o pers.comm). The WMA approach provides opportunities for local communities to benefit from wildlife related activities such as hunting and photographic tourism outside protected areas (URT 1998b). However, the implementation of WMAs has received mixed reactions from local communities in many areas of Tanzania (Nelson et al. 2009) including in the Tarangire ecosystem, where studies have reported that in some areas such as Lake Burunge WMAs have been established without the consent of villagers and therefore create tension with local communities (Igoe and Croucher 2007). Furthermore, in some villages only the local elites benefit from wildlife activities, which is contrary to the objectives of the community based conservation initiatives (Sachedina 2008) and therefore is likely to affect local peoples' support for conservation in the region. Pastoral grazing areas that were selected for this study were outside Lokisale and Simanjiro Game Controlled Areas (Figure 1).

2.1.5 Cultivated Areas

Although rainfall in the Tarangire ecosystem is very erratic (Kahurananga and Silkiluwasha 1997), cultivation of grazing areas has spread rapidly in recent years in the Simanjiro Game Controlled Area, increasing from less than 10% of the total land area in the 1970's to roughly 35% by 1994, with significant conversions continuing today (IRA 2001). In some areas adjacent to the Lokisale Game Controlled Area, cultivated lands now comprise 43.5% of the total land area (IRA 2001). Generally land use change due to expansion of agriculture has shown negative impacts on wildlife migration. Some of the wildlife migration routes have been eliminated and has severely restricted the movement of animals in the ecosystem (Borner 1985, Bolger et al. 2008, Newmark 2008). Farmlands for this study were selected adjacent to the pastoral grazing areas in the Lokisale, Loiborsoit and Emboreet Villages (Figure 1) and the main crops grown in the farms were maize and beans. Only farms that originally were converted from grazing areas were selected for the surveys.

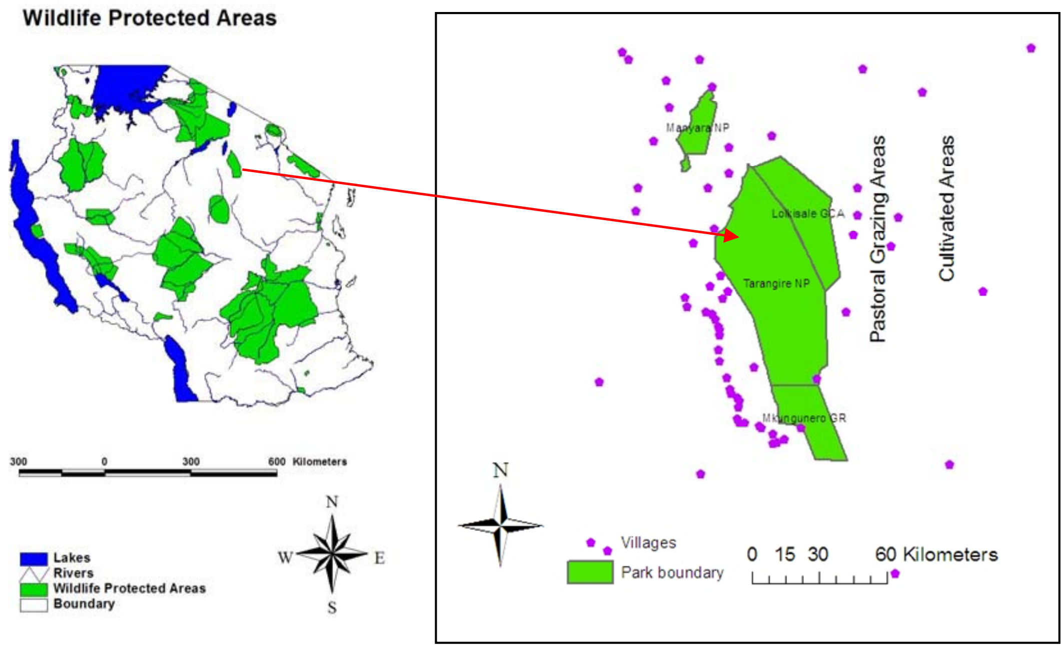

Figure 1: Wildlife Protected areas in Tanzania and the location of the Tarangire ecosystem

2.2 Project Work Schedule

The schedule of field activities was divided into three components. The first component involved introducing the project to the park management in the Tarangire National Park and to village local governments in Lokisale, Loiborsoit and Emboreet villages. The second component involved identifying suitable areas in the park, grazing areas and farmlands outside the park for mammal surveys, including mapping of predifined camera trap locations using Global Positioning Sytems. The second component also involved field surveys using camera traps both during the wet and dry seasons. The third component of the activity schedule involved a month and half of interviews with Maasai agropastoralists. Respondents selected for the interviews were those whose farms were surveyed for mammals. Interviews were only carried out during the dry season. The reason why dry season was selected was that most heads of households (in this case men) were likely to be in the villages during the dry season when members of the families harvest their crops. This bias towards men was partly for logistical reasons but also because it is the local protocol that the household heads (i.e. usually men) need to be approached first before conducting interviews. Women and children usually take a more background role in Maasai families (Kangwana 1993).

2.3 Field Work and Timing

In the Tarangire National Park, field work was carried out from March to May 2006 during the wet season and from June to August 2007 during the dry season. In the grazing areas, field work was carried out from February to March 2007 during the wet season and from August to October during the dry season. Lastly, in the farmlands, field work was carried out from June to August 2006 in the dry season and from March to April 2007 in the wet season. Field surveys were carried out in both dry and wet seasons in order to account for seasonal influences on species richness and abundance, because previous studies showed seasonal movements for some species between the park and pastoral grazing areas (Kahurananga and Silkiluwasha 1997, Gereta et al. 2004). Interviews were conducted between September and October 2007 for a month and half as described above.

2.4 Overview of Methods

Several methods were used for this study due to the interdisciplinary nature of the project. This section therefore provides a general overview of the methods used to determine species diversity, relative and absolute abudances and assessing attitudes of Maasai towards wildlife. Although each of the methods are discussed below, a more detailed account of the methods are provided in each chapter.

2.4.1 Camera Trapping

Data on species diversity and relative and absolute abundances were collected using camera traps. Camera traps are increasingly used in the study of mammals (Tobler et al. 2008) because they perform well under a wide range of environmental conditions and for cryptic and nocturnal species where other methods, such as distance sampling, may fail (Griffiths and van Schaik 1993, Silveira et al. 2003). Camera trap surveys use fixed cameras triggered by infrared sensors, to “trap” images of passing animals. Camera traps are therefore suitable for studying carnivores because carnivores are well camouflaged which makes direct observation difficult, and photographs from camera traps can be easily identified and are verifiable (Cutler and Swann 1999). Camera traps have been used for a wide a range of applications e.g. estimating the abundance of individually recognisable species such as tigers (Karanth et al. 2004a) and leopards (Henschel et al. submitted), estimating density of non-individually recognisable mammals (Rowcliffe et al. 2008), and carrying out species inventories (Silveira et al. 2003). Camera traps have been shown to be more appropriate and accurate for mammal inventories in the open savannas of Emas National Park in Brazil than line transects and track counts (Silveira et al. 2003). Other applications of camera traps include collecting data for estimating species occupancy e.g. sun bears *Herlactos malayanus* (Linkie et al. 2007) and carnivore distribution and habitat use (Linkie et al. 2008, Pettorelli et al. submitted).

However, the use of camera traps can have several problems that may affect data collection. For instance, for this study 29 cameras with films were stolen during the survey outside the park in pastoral grazing areas and farmlands despite reinforcing the cameras with metal boxes and chains and locks. Five more cameras were destroyed by fire in the Tarangire National Park. Exposed film, and hence data, as well as the cameras, were lost in these events. Furthermore, some cameras were also found to take photographs even in the absence of an animal, and it is not clear what triggered these cameras. Details on the limitations for the use of camera traps will be covered in chapter 7. Despite these caveats, the use of camera traps can provide a powerful means for monitoring a wide range of mammals. Specific details of the surveys are described in the relevant chapters.

2.4.2 Attitude Survey

Understanding attitudes of local communities is becoming increasingly important in wildlife conservation due to the impacts of human-wildlife conflicts (Infield and Namara 2001, Walpole and Goodwin 2001, Zimmerman et al. 2005, Dickman 2008). This is particularly important in Africa where human-wildlife conflict is increasing rapidly due to expanding human populations; increasing areas of set-aside for conservation and increasing demands on resources in and around protected areas (Madden 2004, Browne-Nuñez and Jonker 2008). Addressing these conflicts necessitates understanding people’s attitudes to wildlife because attitudes influence the behaviour of people (Fishbein and Abjze 1975), and understanding these attitudes can help provide a better prediction of the response and support of local communities to wildlife conservation (Browne-Nuñez and Jonker 2008). However, in order to understand people’s attitudes it is important to define what actually is being measured to select the appropriate method (Babbie 1995). People may hold attitudes towards a wide variety of objects e.g. natural resources and animals (Fazio 1995), but for the purpose of this study the term ‘attitude’ is used to refer to feelings of people towards wildlife, particularly carnivores.

Several methods are available to measure attitudes depending on how they have been defined e.g. Mordi (1987) used agree/disagree statements to assess public attitudes towards wildlife in Botswana, while Infield (1988) used fixed-response questions to assess in South Africa and Omondi (1994) used open-ended questions to assess human-wildlife conflict in Kenya. However, as with all data collection methods, survey research has its advantages and disadvantages. For example, on one hand qualitative research uses an array of methods that are interactive and seek to describe, translate and come to terms with the meaning of naturally occurring phenomena in the natural world (Aaker et al. 2001, Marshall and Rossman 2006). Qualitative research has the advantage that it provides a better understanding of people and the issues around them because it involves in-depth interviews and observations, it is flexible and thus valuable information, data, ideas and issues can be collected as they emerge. Such research also helps to focus novel questions and formulate hypotheses (Jacobs et al. 1999, Aaker et al. 2001, Marshall and Rossman 2006). However qualitative research has the disadvantage that it is less structured and may require substantial amounts of time to collect and analyse, and thus may be expensive, especially when a large amount of data is needed. Furthermore, because qualitative research is more small-scale and exploratory in nature it is not usually possible to aggregate sufficient data that can be subjected to rigorous statistical analysis. Investigators with only case study data may find it difficult to argue for the validity of their findings (Smith et al. 1991, Bernard 1994, Jacobs et al. 1999). On the other hand quantitative research methods are always numerical and are designed to be objective and hypothesis-driven. The data collected can be aggregated and subjected to statistical analysis and quantitative research is also generalisable and reliable (Smith 1988, Jacobs et al. 1999). Quantitative research has the disadvantage that it is not flexible and thus may provide little understanding of people's actions and the issues that affect them. It also depends on the availability of prior theories and hypotheses (Jacobs et al. 1999).

For this study, data on attitudes of Maasai agropastoralists to wildlife were collected using semi-structured interviews. Semi-structured interviews have been successfully used in many attitude studies. In Tanzania, both Dickman (2008) and Maddox (2003) used them and showed that depredation was not the main cause of livestock loss around Ruaha and Serengeti respectively. Other studies that have used semi-structured interviews successfully include ones examining attitudes of cattle ranchers to jaguars in Brazil (Zimmerman et al. 2005), attitudes of local people to wildlife benefits in Tanzania (Gillingham and Lee 1999); and protected area/people conflicts in India (Maikhuri et al. 2000). Semi-structured interviews are good in that they provide an opportunity for respondents to provide more detailed answers than fully structured questionnaires do, and they are flexible and can allow people to explain their views and therefore help researchers understand the nature of a particular situation (Bernard 1994, Schensul et al. 1999, Hunter and Brehm 2003). However, semi-structured interviews can be time consuming and therefore can be overtaken by events if such surveys are carried out over a long period of time (Bernard 1994). Semi-structured interviews can also be expensive, especially if large amounts of data are needed, and can be biased both by the interviewer and by the respondent's ability to articulate (Glastonbury and MacKean 2004). Interviews that are designed to assess losses to and conflict with wildlife are likely to have particular biases, such as the general exaggeration of losses, lack of accuracy, and the tendency of respondents to overestimate losses caused by more high-profile species compared to less visible, smaller ones (Niskanen 2005). Considerable time is also needed to develop relationships and gain sufficient trust from respondents in order to reveal sensitive information, such as the killing of protected species (Scholte et al. 1999, Bauer and Hari 2001). In order to address these problems, this survey was carried out with the help of a Maasai assistant who interpreted the questions, and about six months was spent in the villages before we carried out any survey, in order to gain trust.

Despite their drawbacks, semi-structured interviews can be used effectively to assess attitudes, and have provided valuable information regarding peoples' perceptions of large carnivores in previous studies (Oli et al. 1994, Marker et al. 2003b, Dickman 2008). The interviews used here (Appendix I) were designed in a similar way to those used by Dickman (2008) to assess attitudes of pastoralists and agro-pastoralists towards wildlife in southern Tanzania. The questionnaire survey design followed the advice of Schensul (1999) such that simpler and less contentious questions were asked towards the start of the interview and more complex or sensitive issues were raised once the confidence of the respondents had been gained.

The household '*olmarei*' was chosen as the sampling unit, following Dickman (2008) where interviews were restricted to one household per boma. Furthermore, for this study the household interview was restricted to the person whose farm had undergone camera-trap surveying for wildlife species (chapters 3-4). At each village, the project was introduced and the purpose of the research was explained to the village government, and it was made clear that the purpose of my research in the villages was for academic needs. The village chairman or Executive Officer provided us a letter to show to the head of the household to be surveyed, and the assistant explained the purpose of our visit and that the investigations were for academic purposes only. As the villages consisted of only one ethnic group, the Maasai, the interviews were predominantly dominated by male respondents. All interviewees were adults ($\geq$ 18 years old).

The majority of interviews were conducted at the household but two interviews had to be conducted elsewhere (at a school and at the village borehole) where the respondents worked as a cook and watchman respectively, because they were unavailable at their households. Interviews were conducted in Kiswahili and took approximately one hour to complete. Key issues covered during the interview included knowledge and identification of local wildlife species found in their farms and in the grazing areas, classification of species depending on how problematic they were considered to be, views towards wildlife and levels of livestock losses attributed to various causes. Further questions were asked on sources of household income, particularly the main and subsidiary sources of income e.g. involvement in photo tourism, trophy hunting, remittance and the relative importance of income from livestock, agriculture and other sources to respondents' livelihoods. Information was also gathered on a range of other variables, such as the level of stock ownership, the relative amounts of stock loss and use, and details of any depredation.

Levels of conflict were assessed in a similar way to Dickman (2008) where respondents were shown picture cards of 30 species (Appendix II) and were asked whether or not they recognized the species, and if they misidentified it then they were told the correct species. If they knew which species it was, they were then asked whether or not it occurred in their farm and in the grazing areas around their household (within a day's walk), they were also asked to classify whether species posed no problem, a small problem or a big problem, and to explain the reasons for any problem. The cards included one picture of a tiger in order to judge respondents' reliability in recognizing local species because there are no tigers in Tanzania and in the study area. Responses were then coded, where 'no problem' = 0, 'small problem' = 1 and 'big problem' = 2, and a mean problem score for all local species was calculated. This score was used as the main index of conflict.

One of the limitations of the method used in this study was a failure to take into account the presence of other people during the interview, as Dickman (2008) found that the presence of other people during the interview had a significant influence on reliability of the answers. Furthermore, it is possible that the principal investigator's presence during the interview influenced answers from respondents, due to the links with a governmental research Institution, although it is not involved directly in conservation activities on the ground like other institutions such as Tanzania National Parks or NGOs such as the African Wildlife Foundation which has long time presence in the area.

2.4.3 Data Analysis

2.4.3.1 Species Richness

Data on carnivores and prey species richness were analysed using non-parametric methods and the programme EstimateS 7.5 (Colwell 2005). Non-parametric species richness estimators are based on capture-recapture models (Colwell and Coddington 1994) and have been shown to perform well in comparative studies when compared to species richness estimators, which are based on extrapolation of species accumulation curves (Hortal et al. 2006). Results of species richness for each land use type were presented as rarefaction curves which show observed and estimated species richness with 95% confidence limits. The pattern of species richness between land use types was presented as charts with associated standard errors. Shannon-Wiener diversity index (*H*) and species evenness (E), which shows relative proportions of each species represented in the study site, were calculated for each land use type.

2.4.3.2 Species Relative Abundances

Species relative abundances were estimated using photographic rates, number of cameras that detected a particular species, the probability of detecting a given species and occupancy, which refers to the proportion of an area occupied by a given species (MacKenzie *et al*, 2002).

2.4.3.3 Photographic Rates.

Species photographic rates for each land use type was calculated as the number of photographs of a given species divided by the total number of camera trap days in that land use type. Total number of trap days were obtained by adding the number of camera trap days from each camera station when the camera was functioning. Photographic rates are useful for providing rapid assessment of species abundance for conservation planning if they are properly calibrated

(Jennelle et al. 2002). This is important because species trapping rates may vary between habitats (Carbone et al. 2001).

2.4.3.4 Number of Cameras that Detected Species

The total number of cameras that detected a given species for each land use type was determined by adding all cameras stations that detected the species during the survey and were plotted against land use types and presented in a chart with associated standard errors at 95% confidence limits. This method, however, has one limitation in that a species may be present in an area but may not be detected e.g. due to vegetation cover and thus abundance of the species is underestimated. To address this problem the probability of detecting a given specis was determined. Details of each of the other methods are described below.

2.4.3.5 Species Detection Probabilities

Generalised Linear Mixed Models in program R (R Core Development Team 2008) were used to investigate effect of land use type and season on the probability of detecting species. Information on species detection or non-detection was used as the the dependent variable and land use type and season as independent variables. The models were weighted by camera trap days in a binomial function and the best fit model was selected based on the lowest Akaike Information Criteria (AIC) that had higher level of significance (Burnham and Anderson 1998). Results are presented with significance level of each land use type and individual species and season and are depicted by ** for $P < 0.01$ and * for $P < 0.05$).

2.4.3.6 Species Occupancy

Species occupancy was determined using the methods described by Mackenzie *et al* (2002). A capture history for each species was developed for the sampling period. The capture history consisted of vectors of '1's and '0's indicating detection and non detection of a given species. In order to estimate species occupancy, a species was asigned '1' for a particular day if it was photographed on that day or '0' if it was not photographed on that day. This information was used to developed a species capture matrix which consisted of vectors for each species with rows representing camera stations and columns representing trapping day. However if there are many non-detections (zeros) in the matrix, it is unlikely that program PRESENCE will compute reliable results. One way to address this is by breaking down the data matrix into longer trapping occasions or intervals that will reduce the number of non-detections e.g. increasing the length of trapping occasion from one to ten days. For this study 10 days interval were used as a trapping occasion and the program PRSENCE (MacKenzie et al. 2002) was used for determining species occupancy. Because not all species present will be detected during the survey, the estimation of occupancy also involved estimating detection probability in order to address the problem of imperfect detection (MacKenzie et al. 2002). Over the last few decade, monitoring species using occupancy methods has been widely adopted, partly because it is relatively easy to quantify occupancy and many ecological phenomena such as species range are naturally expressed in terms of occupancy, but also occupancy is believed to be an informative index of abundance (Royle et al. 2005). There are several monitoring programmes in place that use occupancy. These include large scale monitoring of amphibians in the United States (Hall and Langtimm 2001) and in Switzerland (Pellet and Schmidt 2005) and mammals such as forest elephants (*Loxodanta cyclotis*) in Gabon (Buij et al. 2007).

However, it is important to understand that the use of occupancy to monitor species is based on the assumption that the proportion of an area occupied by a species is positively correlated to the species' abundance i.e. occupancy may increase with increasing abundance (MacKenzie and Nichols 2004). However, it is also equally important to bear in mind that occupancy and abundance are two state variables that address two similar but distinct aspects of a species' population dynamics, i.e. proportion of area occupied by a species in an area and the number of individuals of that species in an area. It is not possible, therefore, to use occupancy to detect changes in species density and, similarly, changes in species occupancy and range may not be detected by monitoring the species abundance (MacKenzie and Nichols 2004). However these discrepancies may be minimal if the species involved is territorial. This is because the number of territorial species in an area may be closely related to the species occupancy if sampling is carried out in an area approximately the same size as the species' territories (MacKenzie and Nichols 2004). However this is often not valid because species territories may overlap. For this study, where non territorial species were involved in the analysis, the objective has been clearly stated in chapter 4.

2.4.3.7 Estimating Species Absolute Abundance

The estimation of absolute abundance in the study area targeted species that can be individually identifiable. Individuals were identified using patterns of spots or stripes on the skin. A photograph of an individual animal was used for comparison with all photographs of the same species obtained during the survey. Photographs showing spot or stripe pattern were identical were grouped together to show that they were from the same individual. Identification of individuals was challenging, in particular for smaller species such as serval because sometimes the animal may be at an awkward position to get a good photograph. In this case, photographs were scanned and magnified in Adobe Photshop program in order to get a position on the animal's skin that could be used to compare with other photographs. Capture-mark-recapture statistical models (White et al. 1982) and the programme CAPTURE (White et al. 1982, Rexstad and Burnham 1991) were used to estimate the abundance. A capture history for each individual was developed. The capture history of an individual consisted of '1's and '0's for each trapping occasion (defined as 10-days intervals of camera trapping). The capture matrices, with rows representing individuals and columns representing trapping occasions, were used in the programme CAPTURE (Otis et al. 1978, White et al. 1982) to estimate species abundance. In order to estimate density, species abundance was divided by the effective sample area, which was obtained by adding a buffer to the camera trap grid equal to half the mean maximum distance moved (½MMDM) among mulitple captures (Wilson and Anderson 1985). The resulting density was expressed as the number of animals per 100 km^2. A more detailed explanation of the procedure is provided in chapter 5.

2.4.3.8 Assessing Attitudes of Maasai Towards Wildlife

Data on attitudes of Maasai towards wildlife were analysed using the Statistical Package for Social Scientist 14.0 (SPSS Inc, Chicago). The one sample Kolmogorov-Smirnov test was used to check for normality of data. Parametric methods were used for data that were normally distributed and non-parametric methods were used for data that were not normally distributed. For continuous variables, the Kruskal-Wallis H test (KW χ^2) was used to compare differences between three variables or groups. Paired sample t-tests were used to compare the means of a variable measured from the same group of people on two different occasions. Other tests used were the Wilcoxon's signed ranks test, to compare variation in non-normally distributed variables between two points in time, and univariate analysis of variance (ANOVA), to compare the mean scores of a continuous variable between two or more groups. Pearson's correlation was used to explore the strength of the relationship between two normally-distributed continuous variables while linear regression was used to determine the predictive ability of a variable on an independent variable e.g. conflict scores.

CHAPTER 3: EFFECTS OF LAND USE ON MAMMAL RICHNESS IN THE TARANGIRE ECOSYSTEM

3.1 Summary

Many wildlife species, especially large carnivores, are threatened by changes in land use practices because their survival depends upon large and intact ecosystems. Such intact habitats are becoming increasingly scarce due to land use change as a result of expanding human populations. The aim of this chapter is to explore the effects of land use practices on carnivore and non-carnivore biodiversity in the Tarangire ecosystem. Camera traps were used to determine species richness in three land use types with increasing human pressure, namely (1) land within the Tarangire National Park, (2) pastoral grazing areas outside the park; and (3) cultivated areas outside the park. Results showed that species detectability, seasonal movements and land use practices appeared to be important factors influencing species richness. Overall there was little difference in carnivore species richness between the park and the pastoral grazing areas. 19 carnivore species were photographed in the park and 17 in the grazing areas. Carnivore richness was lowest in the cultivated areas where only 6 species were detected. Non-carnivore species richness also varied between land use types: it was highest in the pastoral grazing areas, where 22 species were photographed, medium in the park, where 18 species were photographed and lowest in the cultivated areas, where 14 species were photographed. Most large mammals were not found in the cultivated areas, demonstrating that large mammals were more sensitive to environmental change than small mammals. Although the park holds a rich mammal community and is fully protected, pastoral grazing area outside the park also held a significant mammal community and is therefore important for overall conservation of wildlife in the ecosystem. However, the increasing cultivation of pastoral rangelands outside the park is a major threat to wildlife in the ecosystem.

3.2 Introduction

Many species, especially large carnivores, are threatened by loss of habitat due to expanding human population and change in land use practices (Nowell and Jackson 1996, Sillero Zubiri et al. 2004, TAWIRI 2006, 2007a, b, c, e). Consequently, since areas outside reserves are often under pressure from human activities, carnivore conservation to date has focused mainly upon the protected area systems (Linnell et al. 2000). But protected areas may provide sufficient protection mainly for smaller and less widely-ranging species; most protected areas may be too small to sustain viable populations of wide-ranging species, such as large carnivores and some ungulates, in the longer term (Woodroffe and Ginsberg 1998, Ottichilo et al. 2001, Gereta et al. 2004, Thirgood et al. 2004). Therefore successful conservation of wildlife must not only focus upon protected areas but also the surrounding unprotected lands as well (Newmark 1996, Woodroffe 2000b). This is important because many wildlife species occur on these unprotected lands (Homewood and Rodgers 1991, Western and Gichohi 1993). For example, species such as cheetah (Marker-Kraus and Kraus 1994) and wild dog (Creel and Marusha 1996) appear to be the same inside and outside protected areas, probably because of low densities of prey, but also it may be because they occur at low densities and therefore depend on a large landscape of protected and unprotected areas (Durant 1998). Furthermore, human-dominated land outside reserves can also be very important for medium-sized carnivores since many large carnivores cannot survive because of anthropogenic pressures (Woodroffe 1997). The absence of large carnivores in human dominated landscapes may result in a 'mesopredator release' where smaller and medium-sized carnivores benefit from the absence of large competitors (Groom et al. 2006). However, as human populations increase outside protected areas, demand for resources and changes in land use may have significant impact on species.

Despite the lands outside protected areas being important for conservation (Western 1989, Bernnet 1998), little is known about the impact of land use change on carnivores. This is partly because carnivores are often difficult to study directly, due to their low densities, nocturnal nature, and secretive habits (Wilson et al. 1996, Sargeant et al. 1998, Stander 1998) Consequently, the sensitivity of most species to anthropogenic disturbances are poorly understood (Crooks 2002). However, these impacts of anthropogenic disturbances are not specific to carnivores, but also affect non-carnivore species such as ungulates, which are an important group of species in their own right as well as often representing important prey for carnivores (Carbone et al. 1999). Therefore, understanding the impact of human activities on both carnivores and non-carnivores is a necessary component of a comprehensive conservation strategy to cover a diverse range of species in an ecosystem. Both species diversity and richness play an important role in assessing the wildlife conservation potential of areas (Margules and Usher 1981, Baskin 1994), and these metrics are frequently used to judge the success of conservation efforts (Hall and Willig 1994) and prioritisation of protected areas for wildlife (Margules et al. 1988, Travaini et al. 1997). Higher species diversity is generally thought to indicate a more complex and healthier community, because a greater variety of species allows for more species interactions, leading to greater system stability and better environmental conditions (Margules et al. 1988, Travaini et al. 1997).

The aim of this chapter is to examine the effect of land use type on both carnivore and non-carnivore diversity in the Tarangire ecosystem by comparing their relative species richness across three areas, namely within Tarangire National Park, in pastoral grazing areas and in cultivated areas outside the park. This chapter will focus on five key questions.

1. What is the status of carnivore and non-carnivore species richness in the park, pastoral grazing areas and cultivated areas?

2. Is there seasonal variation in species richness within and between land use types?
3. Are some species more sensitive to land use type than others?
4. Is there evidence that land use practices influence interactions between carnivore species leading to mesopredator release?
5. What are the implications of land use change for carnivore conservation?

3.3 Materials and Methods

This study was carried out in the Tarangire ecosystem, specifically within the Tarangire National Park, in pastoral grazing areas outside the park and in cultivated areas outside the park. These areas were located within the Simanjiro and Monduli and Babati districts (Figure 2). Description of the study sites are presented in chapter 2.

3.3.1 Data Collection

Data on species richness were collected using cameras traps. DeerCam 300 passive infrared cameras (Non Typical Inc, Park Falls, USA). The delay between pictures was set to one minute and the sensitivity for the infrared sensor was set to high. Cameras were placed along animal trails at approximately 40 cm above the ground and were left to operate for 24 hours a day for 30 to 75 days. All camera stations were georefenced using Global Positioning System (GPS) in order to be able to locate cameras for replacing films and batteries. Cameras were checked regularly after every 10 days to replace films and batteries and all films were developed and printed and all mammals were identified to species level.

Two field surveys were carried out in 2006 and 2007 as follows: In 2006, from 13th March to 27th May field survey was carried out in the Tarangire National Park during the wet season and from 6th June to 22nd August during the dry season in the cultivated areas outside the park in the

Simanjiro and Monduli disricts. The field surveys in 2006 used a systematic camera trap grid with 20 camera stations per site. Each camera station had double cameras placed opposite each other. In 2007 field surveys were conducted as follows: From 13th February to 19th March during the wet season in the pastoral grazing areas and from 25th March to 30th April during wet season in the cultivated areas. Both surveys were conducted outside the park in the Simanjiro and Monduli districts. Other field surveys were conducted from 12th June to 26th August in the park during the dry season, and from 30th August to 7th October in the pastoral grazing areas outside the park also during the dry season. All four surveys in 2007 used 80 camera stations with a single camera at each station placed at 2 km intervals, except in the park where the survey used double and single cameras as follows: in the same 20 camera trap grid that was surveyed in 2006, camera spacing was reduced from 2 km to 1 km and the number of camera stations were increased from 20 to 42. Also an additional 38 single cameras were placed outside the grid at 2 km interval to provide additional coverage (Figure 2), for estimating species occupancy which will be discuused in chapter 4. The reason for the reduction in camera spacing and increase in the number of cameras was to increase potential for recaptures at different camera stations for estimating absolute abudance, which will be dicussed in chapter 5.

Plate 1: A camera trap in pastoral grazing areas outside the park

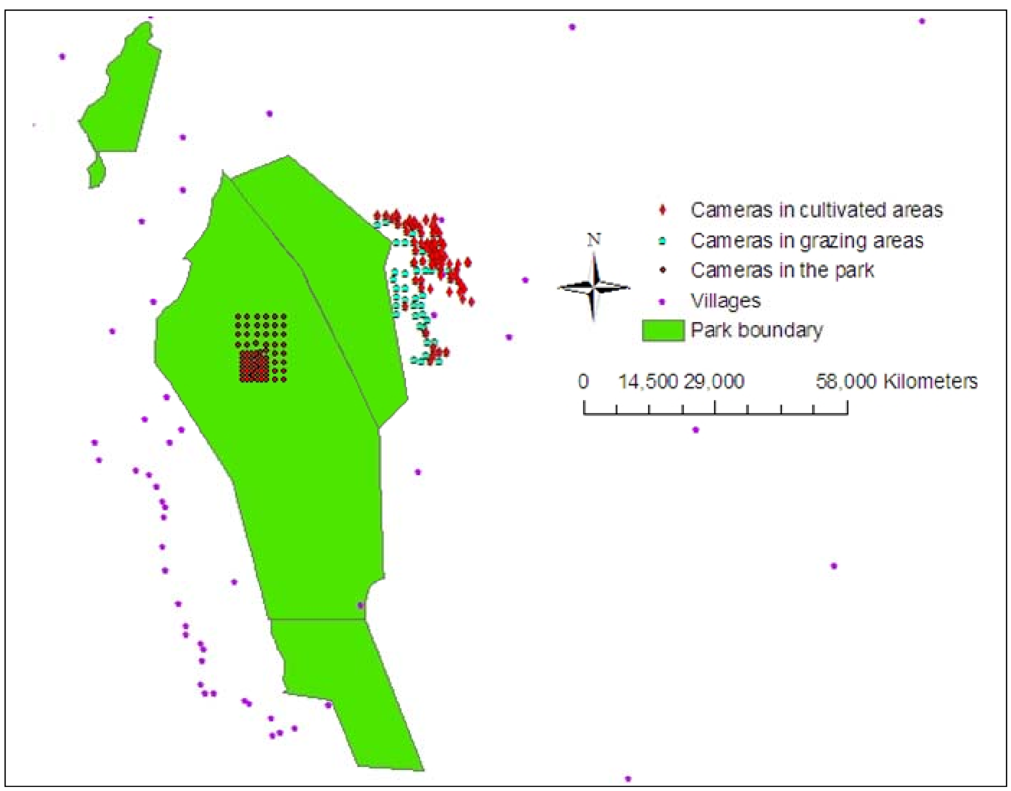

Figure 2: Location of study areas and camera traps in the Tarangire National Park, pastoral grazing areas, and cultivated areas outside the park with first and second survey combined.

3.3.2 Data Analysis

3.3.2.1 Species Richness

Mammal photographs were identified to species level and a species list for each site was compiled. However, not all mammals are equally likely to be detected during any survey, some species that were present during the survey may be missed (Nichols and Conroy 1996, MacKenzie 2002). Therefore when conducting any inventory, it is important to evaluate its completeness in order to estimate how many more species might be detected by further sampling effort (Magurran 1988). This is especially important when comparing species diversity between sites or when investigating changes in species composition over time. Various methods have been developed to assess the completeness of the inventory in each case, and estimate the true number of species in incomplete surveys (Soberón and Lorente 1993, Colwell and Coddington 1994, Koh et al. 2004). These methods can be divided broadly into two classes: (a) species richness estimators based on extrapolation of species accumulation curves; and (b) non-parametric estimators which are related to capture-recapture models (Colwell and Coddington 1994). The latter technique usually performs better in comparative studies (Walther and Moore 2005, Hortal et al. 2006) and hence was used for this study.

The use of camera traps provided information on species presence/absence. The program EstimateS 7.5 (Colwell 2005) was used to estimate species diversity and richness. Program EstimateS has been widely used for estimating species richness for animals and plants and has been shown to provide best estimates (Tøttrup et al. 2005, Bobo et al. 2006, Hortal et al. 2006). For each of the three land use types, the number of carnivores that had been photographed (hereafter referred to as observed species richness) was recorded and the expected species richness (estimated species richness) was determined using Programme EstmateS based on the distribution pattern of observed species richness. In order to achieve this, a presence/absence data matrix was generated for each land use type. The matrix consisted of columns representing species and rows representing trapping occasions, with cells containing the total number of photographs of each species on each occasion. In the data matrix, zero was used where a species was not detected during the trapping occasion and the total number of photographs for that trapping occasion where a species was detected. The programme EstimateS was then used to calculate estimated species richness using the first-order Jackknife that was initially designed to estimate population size from capture-recapture data. This method allows capture probabilities to vary by individuals (Burnham and Overton 1978, Burnham and Overton 1979). The model can equally be applied to estimate species richness (Colwell and Coddington 1994, Boulinier et al. 1998, Chazdon et al. 1998, Nichols et al. 1998, Hughes et al. 2002). The Jackknife estimator is assumed to provide a good estimate of species richness if the proportion of rare species (those which are represented in only one or two samples) is low (Chao 1987, Nichols and Conroy 1996), as was the case for this study. The percentage of observed species richness to estimated species richness (completeness) was also determined in order to compare these two measures.

3.4 Results

3.4.1 Species Distribution between Land Use Types

Some carnivores showed limited distribution patterns. For example, leopard was only photographed in the park, while other large carnivores such as lion and spotted hyaena were photographed both in the park and in the pastoral grazing areas but not in the cultivated areas. Some medium-sized and small carnivores also showed a limited distribution e.g. caracal was photographed only in the park and bushy-tailed mongoose was photographed only in the grazing areas. Other medium-sized carnivores (aardwolf, honey badger and bat-eared fox) were photographed in the park and in the grazing areas but not in the cultivated areas (Table 1). Similarly some non-carnivores also showed limited distribution e.g. springhare was photographed only in the cultivated areas while bush duiker was photographed only in the grazing areas. Hartebeest, buffalo, bushbuck, Thompson's gazelle, waterbuck and olive baboon were photographed only in the park and the grazing areas, while steinbuck and bush pig were photographed only in the grazing areas and cultivated areas (Table 2).

3.4.2. Species Richness between Land Use Types

A total of 19 carnivore species were photographed during the survey. A list of the species photographed is shown in Table 1. In the Tarangire National Park 19 carnivore species were photographed compared to 17 in the pastoral grazing areas and 6 in the cultivated areas. The decrease in species richness was not statistically significant between the park and grazing areas, but was significant both between the park and the cultivated areas and between the grazing areas and cultivated areas (Figure 3). With the exception of the cultivated areas during the wet season, the Jackknife estimators predicted that a number of species may have been missed during the survey. For example, in the most extreme case, in the pastoral grazing areas the Jacknnife estimator predicted that only 65.4% of the estimated carnivore species richness were photographed during the wet season (Table 2). However, even the estimated species richness did not reach an asymptote which implies that additional species would have been photographed with further sampling effort (Figure 6). A similar pattern was also found in the park both during the wet and dry season and in cultivated areas during the dry season, but the rarefaction curves show that estimated species richness based on the sample of observed species richness (Figures 5 and 7 respectively) had started reaching an asymptote. This shows that additional survey effort would not be expected to lead to a significant increase in the number of species photographed.

A total of 23 non-carnivore species were also photographed during the survey (Table 3). Of these, 22 were photographed in the grazing areas, 18 in the park and 14 in the cultivated areas, revealing a significant difference in species richness between land use types (Figure 3). Except in the park during the dry season, the Jackknife species richness estimators again predicted that a number of species may have been missed during the survey. For example, in the park during the wet season only 84.4% of the expected species were photographed (Table 4). Nonethless even the estimated species richness during the wet season (Figure 8) did not reach an asymptote which implies that with additional sampling effort more species would have been photographed. Similarly in the grazing areas both during the wet and dry season observed and estimated species richness (Figure 9) did not reach an asymptote which also implies that there was potential for more species to be photographed with additional sampling effort. This pattern was also observed in the cultivated areas (Figure 10).

Table1: Carnivores photographed in each land use type

Species	Park	Grazing areas	Cultivated areas
Canidae			
Black-backed jackal *Canis mesomeles*	√	√	√
Bat-eared fox *Otocyon megalotis*	√	√	-
Felidae			
Caracal *Felis caracal*	√	-	-
Leopard *Panthera pardus*	√	-	-
Lion *Panthera leo*	√	√	-
Serval *Felis serval*	√	√	√
Wild cat *Felis silvestris*	√	√	√
Herpestidae			
Banded mongoose *Mungos mungo*	√	√	-
Bushy-tailed mongoose *Bdeogale crassicauda*	-	√	-
Dwarf mongoose *Helogale parvula*	√	√	-
Egyptian mongoose *Herpestes ichneumon*	√	√	-
Slender mongoose *Galerella sanguinea*	√	√	-
White-tailed mongoose *Ichneumia albicauda*	√	√	-
Hyaenidae			
Aardwolf *Proteles cristatus*	√	√	-
Spotted hyaena *Crocuta crocuta*	√	√	-
Striped hyaena *Hyaena hyaena*	√	√	√

Mustelidae			
Honey badger *Mellivora capensis*	√	√	-
Zorilla *Ictonyx striatus*	√	√	√
Viverridae			
Common genet *Genetta genetta*	√	√	√
Large spotted genet *Genetta maculata*	√	-	-

Table 2: Observed and estimated carnivore species richness in the Tarangire National Park, pastoral grazing areas and cultivated areas outside the park with standard deviations in brackets.

	Park		Pastoral grazing areas		Cultivated areas	
Parameter	Wet season	Dry season	Wet season	Dry season	Wet season	Dry season
Trap days	1351	3699	1951	1852	2038	783
Cameras	20	75	69	73	70	20
Observed species richness	18	18	11	13	5	6
Jackknife 1 (±SD)	19.97 (±1.39)	18.99 (±0.99)	16.82 (±2.18)	17.87 ±2.06)	5 (±0.12)	6.97 (±0.97)
Shannon-Weaver diversity index	2.4 (±0.01)	2.4 (±0.01)	2.01 (±0.04)	2.1 (±0.02)	1.38 (±0.03)	1.59 (±0.02)
Evenness (E)	1.92	1.95	1.93	1.89	1.97	2.04
% completeness	90.2	94.5	65.4	72.7	100	86.1

Table 3: Non-carnivores photographed in each land use type.

Species	Park	Grazing areas	Cultivated areas
Cercopithecinae			
Olive baboon *Papio anubis*	√	√	-
Vervet monkey *Cercopithecus pygerthrus*	√	√	√
Leporidae			
Cape hare *Lepus capensis*	√	√	√
Pedetidae			
Spring hare *Pedetes capensis*	-	-	√
Hystricidae			
Crested porcupine *Hystix cristata*	√	√	√
Orycteropodidae			
Aardvark *Orycterpus afer*	√	√	√
Elephantidae			
African elephant *Loxodonta africana*	√	√	-
Equidae			
Burchell's zebra *Equus burchelli*	√	√	√
Suidae			
Bush pig *Potamochoerus larvatus*	-	√	√
Warthog *Phacochoerus africanus*	√	√	√
Giraffidae			
Masai giraffe *Giraffa camelopardalis*	√	√	√
Bovinae			
African buffalo *Syncerus caffer*	√	√	-

Bushbuck *Tragelaphus scriptus*	√	√	-
Lesser kudu *Tragelaphus imberdis*	√	√	-
Eland *Taurotrogus oryx*	√	√	√
Antelopinae			
Bush duiker *Sylvicapra grimmia*	-	√	-
Steinbuck *Raphicerus campestris*	-	√	√
Kirk's dikdik *Madoqua kirkii*	√	√	√
Waterbuck *Kobus ellipsiprymnus*	√	√	-
Thomson's gazelle *Gazella rufifrons*	-	√	-
Grants gazelle *Gazelle grantii*	√	√	√
Aepycerotinae			
Impala *Aepyceros melampus*	√	√	√
Alcelaphinae			
Hartebeest *Alcelaphus buselaphus*	√	√	-

Table 4: Observed and estimated non-carnivore species richness in the park, pastoral grazing areas and cultivated areas outside the park with standard deviation in brackets.

	Park		Grazing areas		Cultivated areas	
Parameter	**Wet**	**Dry**	**Wet**	**Dry**	**Wet**	**Dry**
Trap days	1351	3699	1951	1852	2038	783
Cameras	20	75	69	73	70	20
Observed species richness	16	17	20	16	14	12
Jackknife 1 (±SD)	18.96 (±1.69)	17 (±0.05)	21.94 (±1.34)	17.95 (±1.36)	15.96 (1.37)	14.92 (±1.64)
Shannon-Weaver diversity index (H') (±SD)	1.62 (±0.01)	1.9 (±0.01)	1.9 (±0.02)	1.95 (±0.01)	2.07 (±0.01)	2.15 (±0.02)
Evenness (E)	1.35	1.54	1.46	1.63	1.8	1.99
% completeness	84.4	100	91.2	89.1	87.7	80.4

Plate 2: Aardwolf photographed in the park

Plate 3: Steinbuck photographed in the cultivated areas

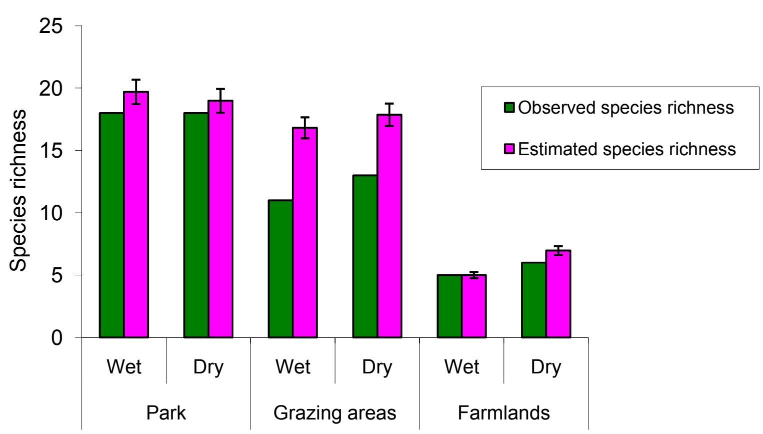

Figure 3: Pattern of observed and estimated carnivore species richness using the first order Jackkinfe in the Tarangire National Park, pastoral grazing areas, and cultivated areas outside the park with error bars representing 95% confidence intervals.

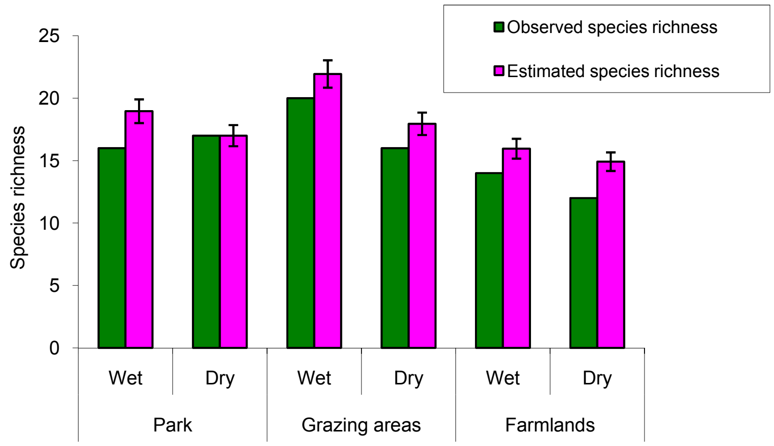

Figure 4: Pattern of observed and estimated non-carnivore species richness in the Tarangire National Park, pastoral grazing areas, and cultivated areas outside the park with error bars representing 95% confidence intervals.

3.4.3 Temporal Variation in Species Richness

Temporal variation was observed for all taxa between wet and dry season within land use types. For the park, 18 carnivore species were photographed during the wet season and 18 during the dry season while in the grazing areas 11 species were photographed during the wet season and 14 during the dry season. In the cultivated areas 5 carnivore species were photographed during the wet season and 6 were photographed during the dry season (Figure 5). However for non-carnivores, 16 non-carnivore species were photographed during the wet season and 17 during the dry season in the park (Figure 6), while in the grazing areas 20 species were photographed during the wet season and 16 in the dry season; and in the cultivated areas 14 species were photographed during the wet season and 12 during the dry season.

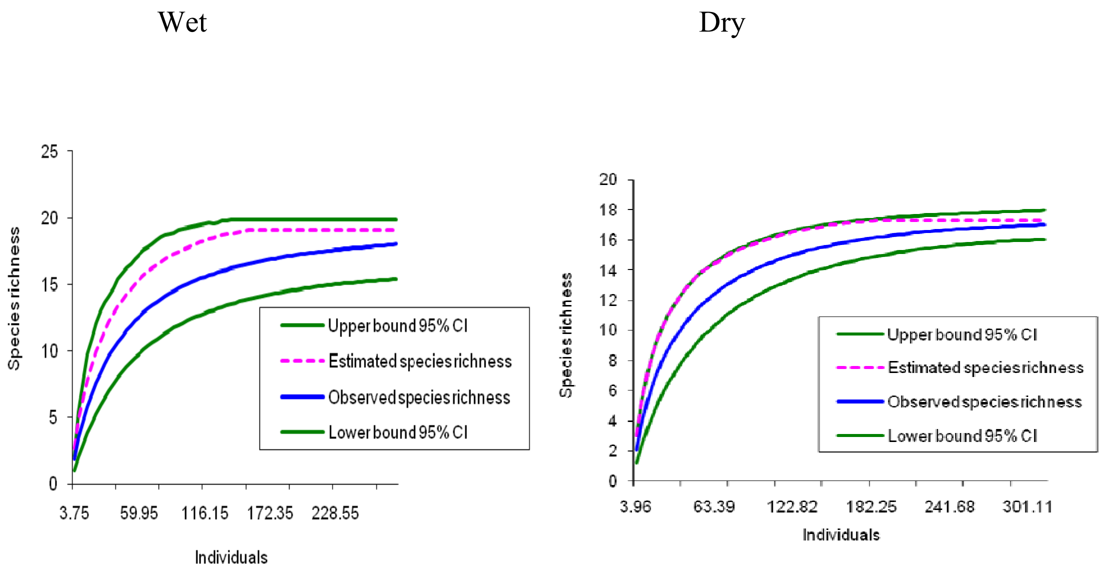

Figure 5: Carnivore species rarefaction curves in the Tarangire National Park under the Jackknife 1 order with 95% confidence interval in the wet season (left) and in the dry season (right).

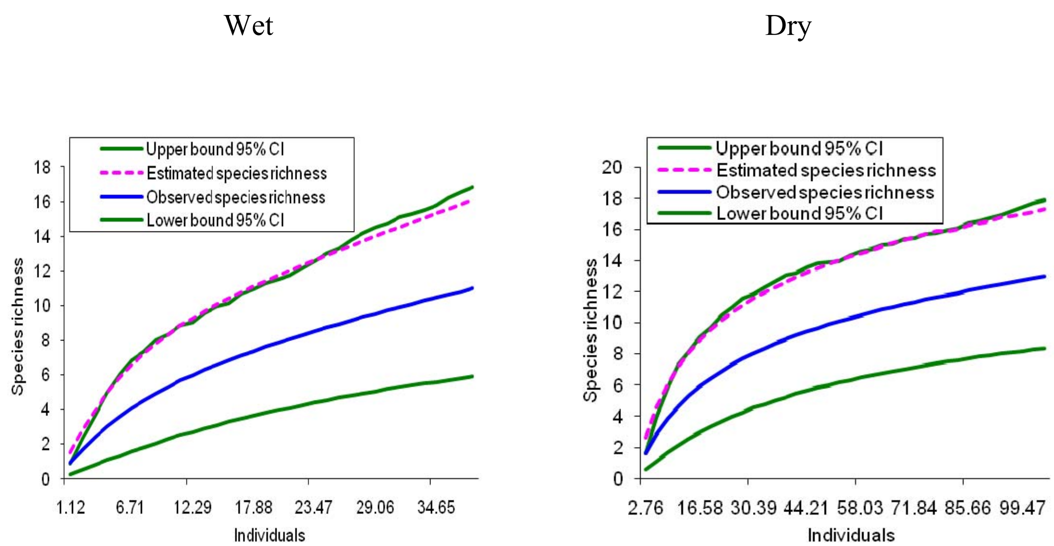

Figure 6: Carnivore species richness rarefaction curves in the pastoral grazing areas under the Jackknife 1 order with 95% confidence intervals in the wet season (left) and in the dry season (right).

Wet Dry

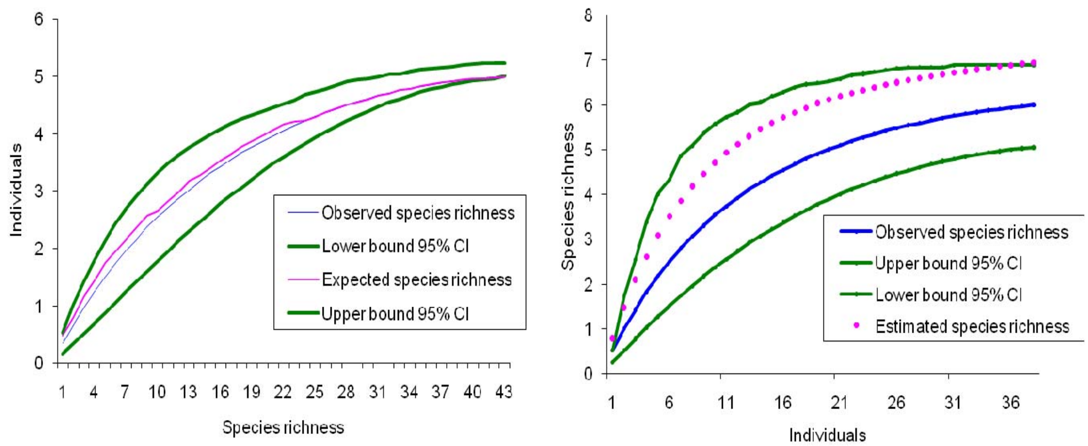

Figure 7: Carnivore species richness rarefaction curves in the cultivated areas outside the park under the Jackknife 1 order with 95% confidence intervals in the wet season (left) and in the dry season (right).

Wet Dry

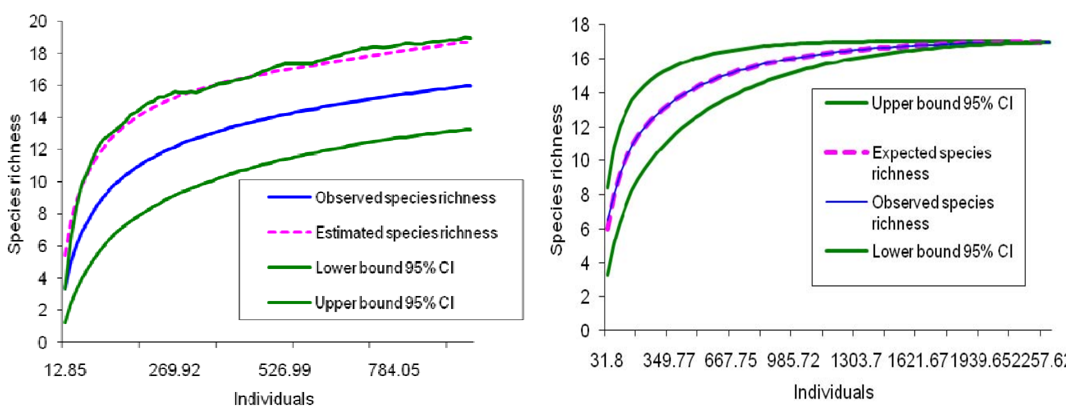

Figure 8: Non-carnivore species rarefaction curves in the Tarangire National Park under the Jackknife 1 order with 95% confidence intervals in the wet season (left) and in the dry season (right).

Wet Dry

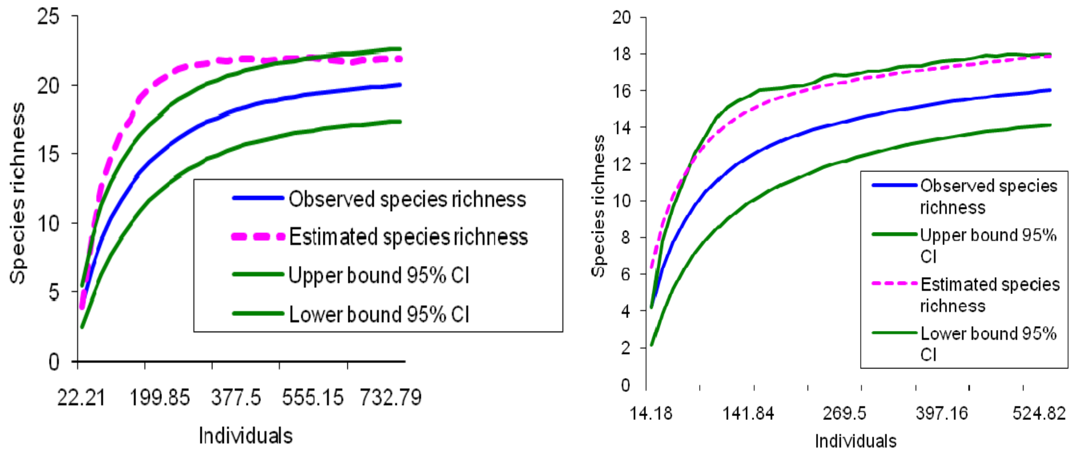

Figure 9: Non-carnivore species richness rarefaction curves in the grazing areas under the Jackknife 1 order with 95% confidence intervals in the wet season (left) and in the dry season (right).

Wet Dry

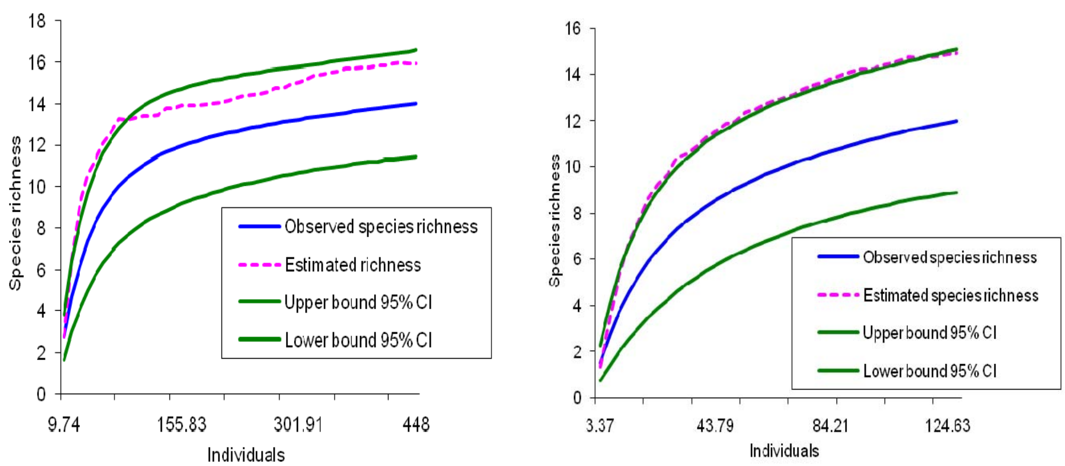

Figure 10: Non-carnivore species richness rarefaction curves in the cultivated areas under the Jackknife 1 order with 95% confidence intervals in the wet season (left) and in the dry season (right).

3.5 Discussion

Results from this study revealed no statistically significant difference in carnivore species richness between the Tarangire National Park and grazing areas, but there was a significant difference between the park and the cultivated areas outside the park. With the exception of striped hyaena, large carnivores were completely absent in the cultivated areas. However, non-carnivore species richness was significantly different between all three land use types. Temporal variation was important for most habitats with some species being detected in one season and not the other.

3.5.1 Difference in Species Richness between Land Use Types

The differences in species richness between land use types shown in this study can be explained by variation in human activities and the intensiveness of use within each land use type. For instance, not all species commonly seen in the Tarangire ecosystem were present in all land use types e.g. aardwolf, banded mongoose, dwarf mongoose, bat-eared fox and honey badger mainly occurred in the national park, and yet are thought to be distributed widely in the region according to their known distribution and ecology (Kingdon 1997, TAWIRI 2006, 2007a, c). However, the presence of bushy-tailed mongoose in the grazing areas was unexpected, although a recent camera trap study in Tanzania by Pettoreli et al. (submitted) found the species in areas not previously recorded, suggesting that the species may be more common than previously thought. Nevertheless, so far all previous records have only been from forests (Kingdon 1997, TAWIRI 2007c, Pettorelli et al. submitted), and therefore the presence of this species in anthropogenically modified savannah ecosystems suggests the species may not be restricted to forests.

It is important to note that the observed variation in carnivore species richness may also be due to variation in trapping effort between land use types. For instance, the percentage completeness in the survey showed that grazing areas had lowest percent completeness i.e. observed species richness was lower than expected richness. This may have been caused by loss of cameras in the grazing areas, particularly during the wet season when some cameras were stolen and therefore some species might have been missed. But also it might be that species were present but were not detected e.g. during the dry season trapping effort was relatively higher and yet completeness was low compared to other land use types.

Higher non-carnivore species richness found in the grazing areas compared to the park or cultivated areas is also attributed to variation in human activities and the intensity of human use between the three land use types. For instance, species that are commonly seen in the Tarangire ecosystem were not present in all land use types e.g. olive baboon, elephant, waterbuck and hartebeest were found in the park and the grazing areas but not in the cultivated areas. Other species which, based on previous reports (Kingdon 1997), were expected to be absent from cultivated areas due to their ecology include buffalo, bushbuck, lesser kudu, hartebeest, bush duiker and Thompson’s gazelle. The significantly higher non-carnivore species richness in pastoral grazing areas than in the park and cultivated areas can be explained by the intermediate disturbance hypothesis (Connell 1978, Huston 1979). The hypothesis states that species diversity will be highest at sites that have had an intermediate frequency of disturbance that prevents competitive exclusion, and will be lower at sites that have experienced very high or very low frequency of disturbance. This is because in some cases disturbance can create a mosaic of habitats which in turn may reduce the effects of interspecific competition among species, and therefore enable coexistence of larger number of ecologically similar species (Fonseca and Robinson 1990, Rosenzweig 1995). In addition, the management of pastoral rangelands often involves grazing, burning and the moderate use of some tree species for fuel or for construction, and it is argued that these management practices play a significant role in maintaining savannah species richness (Homewood and Brockington 1999). Such local use of resources in the savannas may underpin regeneration and foster more habitat diversity and species-rich communities (Fairhead and Leach 1996, Nyerges 1996), which are important for non-carnivores. For example, it has been shown that pastoral land use practices increase frequency of *Acacia tortilis*, which is an important forage crop for both wild ungulates and livestock (Muchiru 1994, Reid and Ellis 1995). Furthermore, high non-carnivore diversity outside protected areas has also been found in abandoned pastoral settlements because of high quality forage produced by heavy dung deposits that enrich soil nutrients (Muchiru et al. 2008). However for this study the observed high diversity of non-carnivores species in pastoral grazing areas may be explained by the disturbance hypothesis as the survey was not conducted in abandoned settlements (bomas).

Finally, it should be noted that non-carnivore prey richness, as determined in this study, is most relevant for large and medium-sized carnivores such as big cats and hyaenas, which rely entirely or largely upon ungulate prey. However, many carnivore diets are varied e.g. aardwolves and bat-eared foxes feed on insects and some canids, mustelids and many viverrids to a great extent subsist on small mammals such as rodents (Gittleman and Harvey 1982, Estes 1991, Carbone et al. 1999). The use of camera traps could not allow sampling of insects and small mammals such as rodents which are important prey for some of these carnivore species.

3.5.2 Temporal Variation in Species Richness

Temporal variation in species presence was observed for most carnivores and non-carnivores. Variation was driven by species that were expected to move seasonally between the park and outside the park in pastoral grazing areas e.g. zebra, buffalo, elephant, Thompson's gazelle and hartebeest move from the park to pastoral grazing areas during the wet season and return during the dry season. Variation also occurred in species thought to be resident e.g. bat-eared fox, black-backed jackal, bush duiker, lesser kudu, bushbuck, cape hare, bush pig, dikdik, impala and steinbuck. This shows that local movements were also important during this study. For example, bat-eared fox and black backed jackal were photographed during the dry season in grazing areas but not during the wet season. Similarly, spring hare and dikdik were photographed during the dry season in the cultivated areas, yet none was photographed during the wet season. The reasons why some species were detected in one season and not the other is not well understood. But, it is possible that a combination of seasonal movements and random variation in seasonal species detectability had an influence on observed temporal variation in species richness. Overall, the observed variation in species richness across the three land use types is likely to be due to a combination of factors such environmental change, seasonal movements and seasonal species detectability.

3.5.3 Species Sensitivity to Land Use Change and Mesopredator Release

This is one of the first studies to demonstrate variation in carnivore species richness along a gradient of changing land use practice in areas of high carnivore diversity. Results show that some species were found in all land use types, suggesting they were less sensitive to environmental change, while other species were only found in some land use types, suggesting that they were more sensitive to environmental change e.g. 13 species were photographed in the park and in pastoral grazing areas but not in the cultivated areas. Additionally nearly half of the carnivores that were not photographed in the cultivated areas or in the pastoral grazing areas were large or medium-bodied species (lion, leopard, spotted hyaena, caracal, honey badger and bat-eared fox). Variation in body size among species has been proposed as an important determinant of extinction probability (Brown 1986, Belovsky 1987, Cardillo et al. 2005), because large species tend to have large home ranges, have lower population densities and are more sensitive to human disturbances (Gittleman and Harvey 1982, Lindstedt et al. 1986, Woodroffe 2000b, Crooks 2002, Kauffman et al. 2007). This is particularly true for large carnivores because they are also dependent on large prey which themselves are vulnerable to habitat change (Gittleman 1989, Carbone et al. 1999). This study found that most large non-carnivore were sensitive to environmental change and tended to be absent in the cultivated areas where the intensity of disturbance is high.

The absence of top predators in an area is thought to influence mesopredator release, a phenomenon that is often characterised by increase in diversity and abundance of medium-sized and small predators (Groom et al. 2006). These results showed no evidence of mesopredator release in cultivated areas where most large carnivores were absent. In fact, some smaller carnivores, particularly viverrids (white-tailed mongoose, banded mongoose, dwarf mongoose, Egyptian mongoose and slender mongoose) were not photographed in cultivated areas but were photographed in the park and the grazing areas. The distribution shown by these species were rather surprising since during the survey we often encountered termitaries, rodents, snakes and beetles which are the prey for such species (Kingdon 1997), this may indicate that these species were sensitive to environmental change. Similar results have been reported in the Arusha National Park where small carnivores avoided human dominated areas including cultivated areas and plantations (Martinoli et al. 2006), although in another study it was found that white-tailed mongoose were common in cultivated areas (Admasu et al. 2004). When compared to other land use types, cultivated areas represent a high level of disturbance and therefore the distribution shown by viverrids may be influenced by intensity of human disturbance, such that only those species less sensitive to environmental change were found in the cultivated areas. This indicates that the mesopredator theory does not hold at a higher level of disturbance since the diversity and abundance of small and medium-sized carnivores also tends to decline.

These results also showed that some non-carnivores were more sensitive to environmental change than others e.g. nine out of the 23 species that were recorded in this study were not found in the cultivated areas, which could be explained by variation in species' ecological needs. For instance, cultivated areas were the only land use type where spring hare was photographed, probably because the species prefers open sandy dry soils and croplands with sufficient cereals (Augustine et al. 1995) which are available in the cultivated areas. Similarly, grazing areas were the only land use type where bush duiker was photographed although this species is thought to be rare in the Tarangire National Park (Charles Foley pers. com). With the exception of eland and zebra, most large and medium bodied non-carnivores tended to be absent from the cultivated areas, which is also attributed to human activities, since large bodied mammals are more vulnerable to local extinction (Brown and Maurer 1986, Belovsky 1987, Cardillo et al. 2005), because they require large and intact areas habitats to survive but also they are hunted outside protected areas because they are potentially most profitable.

Overall, the results of this study demonstrated that some species are more sensitive to environmental change than others. Large and medium bodied species were generally more sensitive to environmental change and tended to be absent in the cultivated areas because of their requirement for large and intact habitats. The absence of large predators does not appear to influence diversity of medium-size and small predators through mesopredator release due to the intensity of human use in cultivated areas.

3.5.5 Implications for Carnivore Conservation

These results showed no significance difference in carnivore species richness between the park and the grazing areas, but did show significant difference between both and cultivated areas thus suggesting that the park and grazing areas outside are important for conserving carnivore biodiversity in the region. Results also showed that large carnivores were more sensitive to envornmental change and were completely absent in the cultivated areas, suggesting further that the presence of large and intact habitats is important for conservation of carnivores especially larger species. Futhermore, higher non-carnivore species richness recorded in the grazing areas shows that carnivores are more sensitive to human activities than their non-carnivore prey, but it also shows that carnivore species richness can be higher in protected areas even when prey species richness is relatively low, which again highlights the importance of protected areas for carnivore conservation.

Finally, the conservation of wildlife in the Tarangire ecosystem faces two major problems. First, the limited size of the protected Tarangire National Park; and secondly increasing human pressure on areas outside the park, particularly in the Simanjiro and Monduli districts to the east of the park, which causes habitat loss and fragmentation due to agricultural expansion. Nevetheless, for the moment the park still holds a rich community of mammals which would otherwise be lost if a substantial part of the ecosystem were cultivated. The impact of this land use change on species abundance will be examined in chapter four. This is important because although measures of species diversity and richness provide useful information for biodiversity conservation planning (Margules and Usher 1981, Magurran 1988, Baskin 1994), they do not reveal the number of individuals of a given species, which is important for monitoring populations. This is because species diversity indices can be high even when general abundance is low (Travaini et al. 1997).

CHAPTER 4: EFFECTS OF LAND USE ON MAMMAL RELATIVE ABUNDANCE IN THE TARANGIRE ECOSYSTEM

4.1 Summary

The aim of this chapter was to investigate whether land use type affect relative abundance of carnivores and their prey in the Tarangire ecosystem. Remote cameras were used to take photographs of mammals in order to estimate relative abundances in three land use types, the Tarangire National Park, pastoral grazing areas and cultivated areas outside the park. Results showed that species photographic rates and the number of cameras that photographed species, the probability of detecting species and species occupancy varied significantly between land use types and were generally higher in the park than in pastoral grazing areas and cultivated areas outside the park. Species relative abundances also varied between seasons. Photographic rates, the number of cameras that photographed species and the probability of detecting species were generally higher during the wet season than during the dry season although there was variation among species. This study highlights the potential importance of camera trap for monitoring the distributions of a wide range of species. It also supports the value of using relative abundance measures in conservation studies. Despite the potential benefits, this work also identifies persistent problems with camera security, particularly outside protected area.

4.2 Introduction

The increasing threats to biodiversity (Smith et al. 2003, Karanth et al. 2006) calls for the need to have effective conservation strategies. In response to these theats to biodiversity, protected areas were established in order to protect key habitas and species diversity (Pressey 1996, Chape et al. 2005) although there has been also other reasons for the establishment of protected areas other than biodiversity conservation e.g protecting areas for religious purposes such as as sacred groove (Chape et al. 2005). However effective management of biodiversity in protected areas requires information on the abundance of species in order to guide management decisions (Maillard et al. 2001, Linkie et al. 2006). Yet to date there is little information available on the abundance of most species not only in protected areas, but also in adjacent unprotected lands which are of critical importance for the long-term maintenance of wildlife populations. This is particularly true for carnivores because estimating abundance especially through direct methods such as distance sampling (Buckland et al. 1993) does not work well because some species are rare, nocturnal or are well camouflaged or occur at low densities and therefore censusing directly such species can be costly and time consuming (Smallwood and Fitzhugh 1993, Ogutu and Dublin 1998). Yet, wildlife managers require rapid and rigourous census methods that can provide them with meaningful and timely results for developing species management strategies (Linkie et al. 2006). One way to achive this can be through the use of indirect methods which can provide index of species abundance such as photographic rates (Carbone et al. 2001) and occupancy and detection probabilities (MacKenzie et al. 2002). In recent years the use of camera traps has become very popular for collecting information on species because the technique has relatively low labour costs, can work well in a wide range of environments where other methods are likely to fail (Karanth and Nichols 1998, O'Brien et al. 2003). The technique is also suitable for cryptic species as is the case of most carnivores. The use of camera traps also provides opportunity to determine indices of species abundances which may provide insights into our understanding of habitat use and species responses to environmental disturbance (Stein et al. 2008)..

The number of photographs taken by cameras per unit time (referred to as the trapping rate or photographic rate (Carbone et al. 2001), contains information about the relative density of a species. Photographic rates have been used widely as an index of abundance to monitor a wide range of species either in combination with other methods or alone. For instance, photographic rates have been used in monitoring diversity and abundance of mammals in Japan (Yasuda 2004) and monitoring species habitat use in Lao, south-east Asia (Johnson et al. 2006). Photographic rates have also been used in combination with distance sampling to monitor species (Silveira et al. 2003). O'Brien et al., (2003) showed that there is a significant correlation between trapping rates and independent estimates of density of species in Sumatra. However, the use of trapping rates as index of abundance requires calibration because trapping rates do not estimate the probability of detecting species which may vary between habitats (MacKenzie et al. 2002, Pollock et al. 2002). One way to address this problem can be through the estimation of species occupancy which provides information on the proportion of area occupied by a species and the probability of detecting the species in the area (MacKenzie et al. 2002). The technique therefore takes into account variation between habitats. Studies have shown that the proportion of area occupied by a species is strongly correlated to the species abundance, so occupancy can be used as a surrogate of abundance (MacKenzie and Nichols 2004). Camera trap photographs have been used successfully to estimate species occupancy, for example, Linkie et al, (2007) used camera trap photographs to estimate occupancy of sunbears in Sumatra. Another approach to address the problem of trapping rates is through the estimation of species absolute abundance based on capture-recapture models (White et al. 1982). However, the use of capture-recapture models is only applicable where individual animals can be identified (Karanth and Nichols 1998, Dillon and Kelly 2007, Henschel et al. submitted), (see also chapter 5) but for the majority of mammals it is not possible to identify individuals and therefore only a fraction of the species can be monitored using this approach. In such circumstances relative abundance estimates are important because they can be used to monitor a wide range of species.

The use of relative abundances may be particularly important in protected areas where human activities are relatively low (IUCN 1994) and therefore relative abundance can provide sufficient information for monitoring species. However for some species and areas absolute abundances are critical (see chapter 5). The aim of this chapter is to examine whether land use type affects the relative abundance of mammals in the Tarangire ecosystem by comparing species relative abundances in the Tarangire National Park, pastoral grazing areas, and cultivated areas. This chapter also explores the potential of using relative abundances from camera trap data for monitoring mammals in the region. In this chapter four questions are examined:

1. Does species relative abundance vary between land use types?
2. Does relative abundance vary between seasons?
3. Is there a potential for using relative abundance from camera trap data to monitor mammals in the Tarangire ecosystem?
4. What are the implications of land use change for wildlife conservation in the region?

4.3 Material and Methods

A description of the study area and background on the use of camera traps to study mammals are presented in chapter 2, so in this section only data collection and analysis methods are described.

4.3.1 Data Collection

Surveys were carried out for a wide range of mammals which could be detected by remote cameras from the size of mongoose to the size of elephants, excluding rodents. Mammal surveys were carried out in 2006 and in 2007. In the 2006 surveys were carried out in the park from 13th March to 27th May during the wet season, and from 6th June to 22nd during the dry season in cultivated areas. Both surveys in 2006 used a systematic grid of camera trapping with 20 camera stations containing double cameras at each station placed at 2 km interval. Details of how camera trap spacing was determined was presented in chapter 3. All cameras were set at approximately 40 cm above the ground in order to take photographs of the species.

In 2007 surveys were carried out as follows: From 13th February to 19th March survey was carried out in the pastoral grazing areas and from 25th March to 30th April 2007 in the cultivated areas during the wet season; and from 12th June to 26th August survey was carried out in the park and from 2nd September to 7th October 2007 in the pastoral grazing areas during the dry season. All surveys in 2007 used 80 camera stations with single camera at each station except in the park where single and double cameras were used for the pupose of estimating species basolute abundance which is discussed in chapter 5. The use of 80 camera stations in 2007 was aimed at increasing spatial coverage for colleting data estimating species occupancy.

4.3.2 Data Analysis

In order to investigate effect of land use type on species relative abundance, camera trap data were analysed at four levels. First, I determined photographic rates, second I determined the number of cameras that photographed each species, third I determined the probability of detecting species; and finally I estimated species occupancy.

4.3.2.1 Species Photographic Rates and Trapping Success

Species photographic rates were obtained by dividing the total number of photographs of a species in a given land use type by the total number of camera trap days in that land use type. The number of trap days varied between landuse types and trapping success was expressed using a standardized measure, the number of photographs per hundred trap days.

4.3.2.2 Number of Cameras that Photographed Species

In each land use type, I determined the total number of cameras that photographed a given species. A species was assigned '1' if it was detected at a particular camera station or '0' if it was not detected at that station. The total number of detections for each species was then plotted against each the land use type and was used an index of species abundance.

4.3.2.3 Probability of Detecting Species

I used Generalised Linear Mixed Models in program R (R Core Development Team 2008) to investigate the probability of detecting a given species across land use types. I modelled detection of species as a function of land use type, species and season. The camera station unique identification number was included as a random effect factor and the model was weighted by camera trap days as effort. The model error structure was binomial. However, there was a large amount of variation in camera trap days between camera stations due to some cameras running out of film or malfunctioned during the survey, therefore camera trap days were standardised by log-transformation to obtain normal distribution as required for parametric analysis.

4.3.2.4 Species Occupancy

Species occupancy analysis is based on data from the park and the pastoral grazing only. Data from the farmland areas were not included because there were too few cameras stations that photographed animals to enable model convergence. Species detection histories were developed whereby a species was assigned '1' if it was detected at a particular camera trapping occasion or '0' if it was not detected. For this study, a trapping occasion was defined as 10 days. A matrix of '1s' and '0s' where rows represented camera placements and columns represented trapping occasions were developed for use in the analysis.

The computer program PRESENCE (MacKenzie 2002) was used for calculating occupancy and detection probabilities. The program uses a likelihood based method built within a mark-recapture statistical framework for estimating the number of sites occupied by a species when the species detection probabilities are < 1. All estimation models assume that (a) sites that are occupied by the species of interest remain occupied for the duration of the survey, (b) that species are not detected when absent, and species may or may not be detected when present; and (c) detecting species at one site is independent of detecting a species at all other sites (MacKenzie 2002). Parameters estimated by the program PRESENCE include ψ a symbol for a Greek word 'Psi' and the probability (p) that a species is present at a site.

A total of 16 species out of 43 photographed were selected for estimating occupancy because they had relatively high number of cameras that detected the species. Of the 16, eight were carnivores (aardwolf, banded mongoose, black-backed jackal, bat-eared fox, common genet, lion, spotted hyaena and white-tailed mongoose) and eight non-carnivores (aardvark, elephant, giraffe, impala, dikdik, warthog, buffalo and zebra). However, lion, spotted hyaena, giraffe, zebra, elephant and buffalo range widely and therefore were expected to violate the assumption that detecting a species at one site is independ of decting at all other sites (MacKenzie 2002). The inclusion of these species in the analysis was intended to determine habitat selection or use rather than occupancy. Elephants were also included in the analysis because of the role they play in habitat modification (Birkett 2002) which in turn may affect carnivore occupancy. Throughout the remaining chapter, I refer to 'habitat use' rather than occupancy for wide ranging species mentioned above.

To estimate occupancy, a set of four models were develeped which might explain species occupancy and detection probabilities in the study area. In the first model, I assumed that occupancy and detection probability were constant in all study sites (ψ (site) p()). In the second model, I assumed that occupancy varies between study site but detection probability remains constant (ψ (site) p()). In the third model, I assumed that occupancy remains constant but detection probability varies between sites (ψ () p(site)); and in the fourth model, I assumed that both occupancy and detection probability vary between sites (ψ (site) p(site)).

4.4 Results

4.4.1 Photographic Rates, Number of Cameras that Detected Species and Probability of Detecting Species

4.4.1.1 Carnivores

Overall land use type and season had a significant influence on relative abundance of most species. Photographic rates (Table 5) and the number of cameras that photographed species (Figure 11) were generally higher in the park than in pastoral grazing areas and cultivated areas. The probabiliy of detecting carnivores was higher in the park than in the pastoral grazing areas and cultivated areas and the probability was generally higher during the wet season than during the dry season (Table 7). Comparison of photographic rates for large carnivores between land use types showed that photographic rates were generally higher in the park for most species than outside the park in pastoral grazing areas and most species except striped hyaena were not photographed in cultivated areas.

4.4.1.2 Non-carnivores

Land use type also showed significant influence on non-carnivore photographic rates (Table 6) and the number of cameras that photographed species (Figure 12) and the probability of detecting species (Table 8). Results show that photographic rates, number of cameras that photographed species and the probability of detecting non-carnivores were generally higher in the park than in the pastoral grazing areas and cultivated areas and were particularly higher during the wet season than during the dry season. Photographic rates and the number of cameras that photographed large non-carnivores were higher in the park than outside the park in the grazing areas with some species, such as elephants, completely being absent in the cultivated areas.

Table 5: Carnivore photographic rates (number of photos per trap night) in the Tarangire National Park, pastoral grazing areas, and cultivated areas outside the park.

Species	Park		Pastoral grazing areas		Cultivated areas	
	Wet	Dry	Wet	Dry	Wet	Dry
Aardwolf	0.0215	0.0019	0.0010	0.0076	-	-
Banded mongoose	0.0318	0.0234		0.0151	-	-
Bat-eared fox	0.0030	0.0035	-	0.0049	-	-
Black-backed jackal	0.0192	0.0185	-	0.0181	0.0015	0.0046
Bushy-tailed mongoose	-	-	0.0179	-	-	-
Caracal	0.0015	0.0005	-	-	-	-
Common genet	0.0489	0.0057	0.0036	0.0027	0.0049	0.0114
Dwarf mongoose	0.0015	0.0024	0.0015	0.0020	-	-
Egyptian mongoose	0.0007	0.0005	0.0001	0.0001	-	-
Honey badger	0.0044	0.0049	-	0.0011	-	-
Larges spotted genet	-	0.0002	-	-	-	-
Leopard	0.0104	0.0008	-	-	-	-
Lion	0.0215	0.0030	0.0041	-	-	-
Serval	0.0126	0.0038	0.0021	0.0001	-	0.0068
Slender mongoose	0.0037	0.0019	0.0015	-	-	-
Spotted hyaena	0.0133	0.0019	0.0056	-	-	-
Striped hyaena	0.0010	-	-	0.0010	0.0100	-
White-tailed mongoose	0.0163	0.0138	0.0001	0.0059	-	0.0011
Wild cat	0.0096	0.0011	-	0.0005	0.0015	0.0068
Zorilla	0.0030	0.0011	0.0001	-	0.0010	0.0023
Number of cameras	20	75	69	73	70	20
Number of trap days	1351	3699	1951	1852	2038	878
Number of carnivore photographs	302	313	40	87	20	29
Trapping success	3.02	3.13	0.40	0.87	0.20	0.29

Table 6: Non-carnivore photographic rates and success in the Tarangire National Park, pastoral grazing areas, and cultivated areas outside the park.

Species & Parameter	Park		Pastoral grazing areas		Cultivated areas	
	Wet	Dry	Wet	Dry	Wet	Dry
Aardvark	0.0170	0.0158	-	0.0086	0.0029	0.0057
African buffalo	-	0.0275	0.0031	0.0130	-	-
African elephant	0.0688	0.0720	0.0010	0.0092	-	-
Burchell's zebra	-	0.0638	0.0041	0.0016	0.0182	0.0023
Bush duiker	-	-	0.0133	0.0016	-	-
Bush pig	-	-	-	-	0.0010	0.0011
Bushbuck	0.0141	0.0038	0.0036	0.0027	-	-
Cape hare	0.0118	0.0089	-	0.0086	0.0093	0.0387
Crested porcupine	0.0089	-	0.0005	0.0059	0.0079	0.0182
Eland	0.0015	0.0022	0.0056	0.0011	0.0128	-
Grants gazelle	0.0015	0.0027	0.0036	0.0011	0.0015	0.0137
Hartebeest	0.0022	0.0097	-	0.0081	-	-
Impala	0.2354	0.2025	0.0010	0.1080	0.0191	0.0034
Kirk's dik dik	0.2953	0.2341	0.0005	0.0664	0.0079	0.0216
Lesser kudu	0.0096	0.0081	0.0005	0.0070	-	-
Maasai giraffe	0.0407	0.0419	0.0056	0.0432	0.0147	0.0023
Olive baboon	-	0.0008	-	0.0005	-	-
Spring hare	-	-	-	-	-	0.0057
Steinbuck	-	-	0.0021	-	0.0064	0.0114
Thomson's gazelle	-	-	0.0282	-	-	-
Vervet monkey	0.0015	0.0008	0.0010	0.0022	0.0034	0.0080
Warthog	0.0252	0.0195	0.0630	0.0027	0.0059	0.0034
Waterbuck	0.0007	0.0019	0.0026	0.0016	-	-
Number of cameras	20	75	69	73	70	20
Number of trap days	1351	3699	1951	1852	2038	878
Number of photographs	993	2643	525	575	228	112
Trapping success (no. photos/100 trap days)	9.93	26.43	5.25	5.75	2.28	1.12

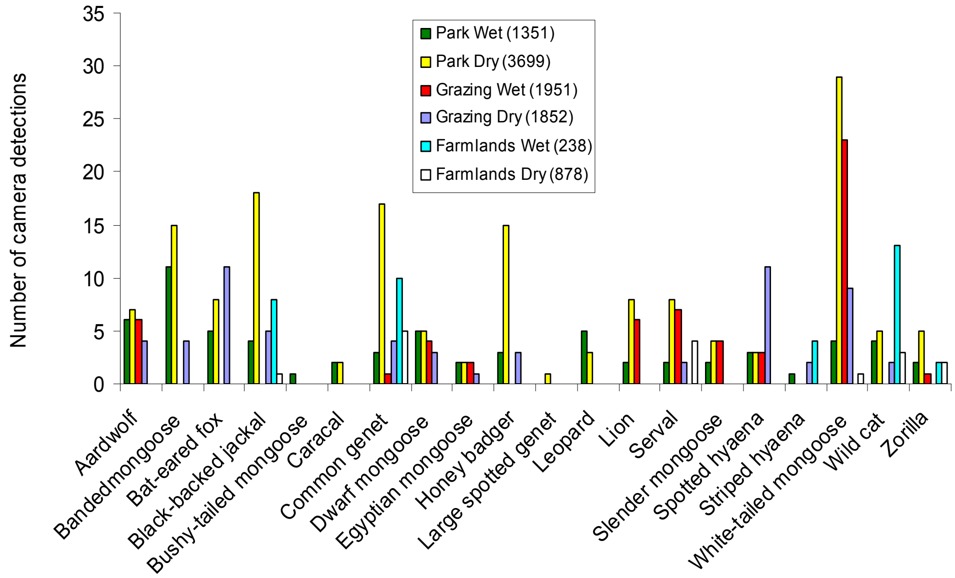

Figure 11: Number of cameras that photographed carnivores in the park, pastoral grazing areas and cultivated areas with camera trap effort in brackets.

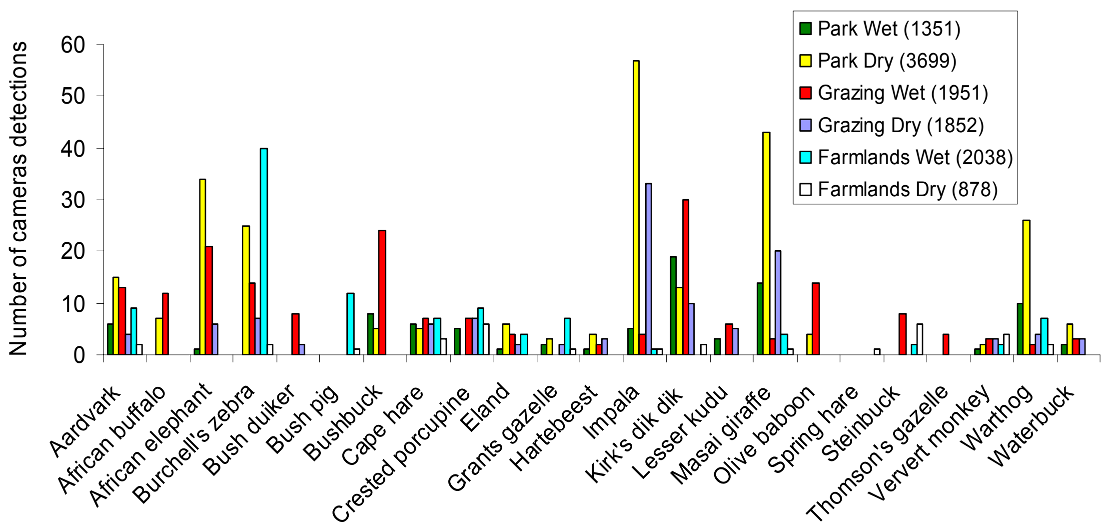

Figure 12: Number of cameras that photographed non-carnivores in the park, pastoral grazing areas and cultivated areas outside the park with camera trap effort in brackets

Table 7: Summary statistics of Generalised Linear Mixed Model for carnivore detectability in the park, pastoral grazing areas and cultivated areas where intercept refers to park and site 2 and 3 pastoral grazing areas and cultivated areas respectively.

Parameter	Estimate	SE	z	P
Park (Intercept)	-2.82	0.19	-14.98	<0.001
Pastoral areas - Park	-0.63	0.19	-3.36	<0.001
Cultivated areas- Park	-1.38	0.20	-7.06	<0.001
Wet-Dry	1.50	0.09	16.84	<0.001
Banded mongoose	0.34	0.18	1.85	0.064
Bat-eared fox	-0.22	0.20	-1.103	0.270
Black-backed jackal	0.24	0.19	1.29	0.198
Bushy-tailed mongoose	-2.99	0.48	-6.22	<0.001
Caracal	-2.14	0.34	-6.22	<0.001
Common genet	0.43	0.18	2.35	0.019
Dwarf mongoose	-0.70	0.22	-3.12	0.002
Egyptian mongoose	-1.58	0.28	-5.56	<0.001
Honey badger	-0.50	0.21	-2.32	0.021
Large spotted genet	-3.10	0.50	-6.16	<0.001
Leopard	-1.29	0.26	-4.96	<0.001
Lion	1.07	0.17	6.24	<0.001
Serval	-0.20	0.20	-0.98	0.326
Slender mongoose	0.56	0.18	3.14	0.002
Spotted hyaena	-0.52	0.21	-2.41	0.016
Striped hyaena	-1.49	0.28	-5.35	<0.001
White-tailed mongoose	1.26	0.17	7.49	<0.001
Wild cat	-0.17	0.20	-0.85	0.395
Zorilla	1.10	0.17	6.42	<0.001

Table 8: Summary statistics of Generalised Linear Mixed Model for non-carnivore detectability in the park, pastoral grazing areas and cultivated areas. Intercept refers to park and aardvark and site 2 and 3 refers to pastoral grazing areas and cultivated areas respectively.

Parameter	Estimates	SE	z	*P*
Intercept	-1.38	0.15	-9.10	<0.001
Site 2-Site 1	-0.79	0.13	-6.26	<0.001
Site 3-Site 1	-1.40	0.14	-9.75	<0.001
Wet season-Dry season	0.42	0.07	5.70	<0.001
African buffalo	-1.28	0.24	-5.26	<0.001
African elephant	0.94	0.16	5.73	<0.001
Burchell's zebra	0.75	0.17	4.54	<0.001
Bush duiker	-1.91	0.30	-6.32	<0.001
Bush pig	-1.47	0.26	-5.70	<0.001
Bushbuck	-0.38	0.19	-1.98	0.05
Cape hare	-0.38	0.19	-1.96	0.05
Crested porcupine	-0.19	0.19	-1.00	0.32
Eland	-1.08	0.23	-4.73	<0.001
Grants gazelle	-1.39	0.25	-5.53	<0.001
Hartebeest	-1.61	0.27	-5.95	<0.001
Impala	1.09	0.16	6.70	<0.001
Kirk's dikdik	1.62	0.16	10.16	<0.001
Lesser kudu	0.17	0.18	0.94	0.35
Maasai giraffe	0.89	0.16	5.45	<0.001
Olive baboon	-1.30	0.24	-5.31	<0.001
Spring hare	-4.13	0.77	-5.35	<0.001
Steinbuck	-1.13	0.25	-4.59	<0.001
Thomson's gazelle	-2.86	0.45	-6.41	<0.001
Ververt monkey	-1.39	0.25	-5.52	<0.001
Warthog	0.11	0.18	0.63	0.53
Waterbuck	-1.78	0.29	-6.19	<0.001

4.4.2 Species Occupancy and Habitat Use

Significant effects of land use type were obtained for 7 species out of the 16 that were assessed (Table 9). Species occupancy and habitat use were explained by two most parsimonious models i.e. top ranked models with fewest parameters. These models were ψ (site) p(site) which implies that species occupancy or habitat use and detection probability varied between land use types and ψ (site) p() which implies that species occupancy varied between land use types but detection probability remained constant. Generally species detection probabilities were different between the park and pastoral grazing. Some species had relatively higher detection probabilities in the pastoral grazing areas than in the park which may suggest that some species are widely distributed but are locally less abundant where they occur. But it is also possible that other factors may also influence detection e.g. habitat and physical conditions such as topography.

4.4.2.1 Carnivores

Two species (banded mongoose and black-backed jackal) showed significant results on effects of land use type on species occupancy. No significant results were obtained for lion, spotted hyaena, bat-eared fox, and aardwolf (Table 9). It is possible that data were not sufficient for these species to show significance difference between land use types. For example, although these species were detected in the park and in the pastoral grazing areas, the number of cameras that detected these species was generally lower compared to banded mongoose and black-backed jackal (Figure 11). The top ranked model that explained occupancy for banded mongoose was (ψ (site) *p*(site)) which shows that banded mongoose occupancy and detection probability varied between land use types. Occupancy was higher in the park (ψ ±SE = 0.50 ±0.186, *p* =0.08) than in the pastoral grazing areas (ψ ±SE = 0.06 ±0.032, *p* = 0.26) which suggest that the species was relatively more abundant in the park than in the pastoral grazing areas. For black-backed jackal the top ranked model that explained the species occupancy was also (ψ (site) *p*(site)) which also suggest that occupancy and detection probability varied between land use types. The species occupancy was higher in the park (ψ ±SE = 0.40±0.101, p = 0.12) than in the pastoral grazing areas (ψ ±SE = 0.04 ±0.025, p = 0.44) again suggesting that the species was more abundant in the park than in pastoral grazing areas. Comparison between the two species shows that banded mongooses were relatively more abundant than black-backed jackal in the two land use types.

4.4.2.2 Non-carnivores

Significant effects of land use type on species occupancy were found for five species, zebra, aardvark, giraffe, impala and warthog and non significant results were found for dikdik, elephant and buffalo (Table 9). The lack of significance for dikdik, elephant and buffalo was surprising because the number of cameras that detected the species (Figure 12) varied between land use types but was not statistically different. It is possible that this may be due to low number of cameras that detected these species to allow model convergence. Habitat use for zebra was explained by the model (ψ (site) p(site)) which shows that habitat use and detection probability varied between land use types. The species habitat use was higher in the park ($\psi \pm SE = 0.37 \pm 0.07$, p = 0.29) than in pastoral grazing areas outside the park ($\psi \pm SE = 0.15 \pm 0.03$, p = 0.69) which indicate that the species was more abundant in the park than outside the park in pastoral grazing areas.

For aardvark the top ranked model which explained the species occupancy was (ψ (site) p(site)) which implies that the species occupancy and detection probability varied between land use types. Occupancy for aardvark in the park was higher ($\psi \pm SE = 0.72 \pm 0.299$, p = 0.10) than in pastoral grazing areas ($\psi \pm SE = 0.04 \pm 0.017$, p = 0.07) which indicates that the species was relatively more abundant in the park than in pastoral grazing areas outside the park. The top ranked model which explained habitat use by giraffe was (ψ (site) p()) which shows that habitat use varied between land use type but detection probability remained constant in each land use type. Occupancy was higher in the park ($\psi \pm SE = 0.85 \pm 0.010$) than in the pastoral grazing areas ($\psi \pm SE$, 0.57 ± 0.102) and overall constant detection probability ($p = 0.26$). These results indicate that although detection probability was constant in the park and in the grazing areas, the species was more abundant in the park than in the pastoral grazing areas.

For impala the top ranked model that explained the species occupancy was (ψ (site) p(site)) which means that the species occupancy and detection probability varied between land use types. Occupancy was higher in the park (ψ ±SE = 0.96 ±0.04, p = 0.51) than in pastoral grazing areas (ψ ±SE = 0.61 ± 0.09, p = 0.34), which suggest that impala were more abundant in the park than in pastoral grazing areas. Similarly for aardvark the top ranked model that explained the species occupancy was also (ψ (site) p(site)) which means that the species occupancy and detection probability varied between the land use types. Occupancy for the species was higher in the park (ψ ±SE = 0.67 ±0.120, p = 0.21) in the park (ψ ±SE = 0.06 ± 0.02, p = 0.36) than in pastoral grazing areas. This indicates that warthog were more abundant in the park than in the pastoral grazing areas.

Table 9: Model selection and parameter estimation for occupancy for each species with sufficient data showing best model AIC and (ΔAIC), AIC model weights (*w*), twice log likelihood (-2l) and number of parameters in the model (Npar). Overall best-fit models for each species are shown in bold.

Model, by species	AIC	ΔAIC	*w*	-2l	Npar
Carnivores					
Banded mongoose					
ψ (site) p(site)	**261.60**	**0.00**	**0.495**	**253.59**	**4**
ψ (site) *p*()	261.89	0.29	0.429	255.88	3
ψ () *p*(site)	266.13	4.53	0.051	260.13	3
ψ () *p*()	267.61	6.01	0.025	263.60	2
Black-backed jackal					
ψ (site) p(site)	**296.06**	**0.00**	**0.879**	**288.06**	**4**
ψ (site) *p*()	300.09	4.03	0.117	294.09	3
ψ () *p*()	308.11	12.05	0.002	304.11	2
ψ() *p*(site)	308.75	12.69	0.002	302.75	3
Bat-eared fox					
ψ() p(site)	**258.79**	**0.00**	**0.429**	**252.79**	**3**
ψ () *p*()	259.76	0.97	0.264	255.76	2
ψ (site) p(site)	260.72	1.93	0.164	252.72	4
ψ (site) *p*()	260.99	2.20	0.143	254.99	3
Common genet					
ψ() *p*(site)	**421.49**	**0.00**	**0.622**	**415.49**	**3**
ψ (site) p(site)	423.01	1.52	0.291	415.01	4
ψ (site) *p*()	425.41	3.92	0.087	419.41	3
ψ () *p*()	439.47	17.98	0.000	435.47	2
Spotted hyaena					
ψ () *p*()	**272.53**	**0.00**	**0.508**	**268.53**	**2**
ψ (site) *p*()	274.29	1.76	0.211	268.29	3
ψ() *p*(site)	274.59	1.99	0.188	268.52	3
ψ (site) p(site)	275.93	3.40	0.093	276.93	4
Aardwolf					
ψ () *p*()	**277.87**	**0.00**	**0.443**	**273.83**	**2**
ψ() *p*(site)	279.11	1.24	0.238	273.11	3
ψ (site) *p*()	279.20	1.33	0.278	273.20	3
ψ (site) p(site)	281.04	3.17	0.091	273.04	4
Lion					
ψ() *p*(site)	**196.48**	**0.00**	**0.534**	**192.48**	**2**
ψ (site) *p*()	198.48	2.00	0.197	192.48	3
ψ() *p*()	198.48	2.00	0.197	192.48	3
ψ (site) *p*(site)	200.48	4.00	0.072	192.47	4
White-tailed mongoose					
ψ() *p*(site)	**464.04**	**0.00**	**0.452**	**470.04**	**3**
ψ (site) *p*()	476.51	0.47	0.357	470.51	3
ψ (site) *p*(site)	477.77	1.73	0.190	469.77	4
ψ() *p*()	488.29	12.25	0.001	484.29	2
Non-carnivores					
Aardvark					
ψ (site) *p*(site)	**264.45**	**0.00**	**0.981**	**256.45**	**4**
ψ (site) *p*()	272.38	7.93	0.019	266.38	3
ψ () *p*(site)	281.68	17.23	0.002	277.68	2
ψ () *p*()	279.19	14.74	0.001	273.19	3
Giraffe					

ψ (site) *p*()	**624.01**	**0.00**	**0.445**	**618.01**	**3**
ψ () *p*(site)	624.71	0.75	0.308	618.76	3
ψ (site) *p*(site)	625.99	1.98	0.668	617.99	4
ψ () *p*()	627.56	3.55	0.076	623.55	2
Impala					
ψ (site) p(site)	**730.23**	**0.00**	**0.497**	**722.23**	**4**
ψ () *p*(site)	731.07	0.84	0.326	725.07	3
ψ (site) *p*()	232.29	2.06	0.177	725.29	3
ψ () *p*()	769.09	38.86	0.000	765.09	2
Warthog					
ψ (site) *p*(site)	**387.82**	**0.00**	**0.905**	**379.83**	**4**
ψ (site) *p*()	392.34	4.52	0.095	383.34	3
ψ () *p*(site)	403.24	15.42	0.004	297.24	**3**
ψ () *p*()	425.02	27.20	0.000	411.02	2
Zebra					
ψ (site) *p*(site)	**389.92**	**0.00**	**0.935**	**381.91**	**4**
ψ () *p*(site)	397.03	7.11	0.026	391.03	3
ψ (site) *p*()	397.64	7.72	0.020	391.63	3
ψ () *p*()	397.70	7.78	0.019	391.70	2
Buffalo					
ψ () *p*()	**232.48**	**0.00**	**0.438**	**228.48**	**2**
ψ (site) *p*()	233.41	0.93	0.275	227.41	3
ψ () *p*(site)	234.19	1.71	0.186	228.19	3
ψ (site) *p*(site)	235.40	2.92	0.102	227.40	4
Dikdik					
ψ () *p* (site)	**767.84**	**0.00**	**0.647**	**761.84**	**3**
ψ (site) *p*(site)	769.04	1.22	0.352	761.06	4
ψ (site) *p*()	781.82	13.98	0.001	775.82	3
ψ () *p*()	841.58	73.74	0.000	837.58	2
Elephant					
ψ () *p* (site)	**505.58**	**0.00**	**0.633**	**499.58**	**3**
ψ (site) *p*(site)	507.50	1.92	0.242	499.50	4
ψ (site) *p*()	508.83	3.25	0.125	502.83	3
ψ () *p*()	561.57	55.99	0.000	557.57	2

4. 5 Discussion

These analyses represent the first attempt to investigate variation in relative abundances of carnivores and their prey in different land use types in areas with high diversity of carnivores. My results showed that: (1) relative abundances of most species were higher in the park than in the pastoral grazing areas and cultivated areas outside the park, (2) relative abundances of species varied between seasons, and (3) the use of relative abundances from camera trap data showed a potential for monitoring a wide range of mammals in the Tarangire ecosystem.

4.5.1 Photographic Rates, Number of Camera that Detected Species and Probability of Detecting Species

These results show that for most species photographic rates and the number of cameras that photographed species and the probability of detecting species were generally higher in the park than in the pastoral grazing areas and cultivated areas outside the park, suggesting that most species were more abundant in the park than in the pastoral grazing areas and cultivated areas. The data also showed that temporal variation was important for most species. Species relative abundances varied between seasons. Generally most species relative abundances were higher during the wet season than during the dry season.

4.5.1.1 Carnivores

The observation that photographic rates, number of cameras that photographed species and probability of detecting carnivores were higher in the park than in the grazing and cultivated areas is not surprising. This is because national parks are under strict protection and exclude human activities except for scientific research or photographic tourism. The exclusion of people from protected areas makes protected areas very important for providing key habitats for maintaining wildlife populations, which otherwise would not be possible outside protected areas. This is due to increasing human population and demand for land and other resources which may lead to change in habitat conditions and therefore affect abundance of species. Such anthropogenic disturbances are particularly severe to large carnivores because they are sensitive to change in environmental conditions due to their wide ranging behaviour and their requirement for large and intact habitats to survive (Sillero-Zubiri and Laurenson 2001, Ray 2005). In the Tarangire ecosystem large areas of pastoral grazing areas which are important for wildlife outside the park are increasingly being converted to agriculture and therefore most carnivores especially large ones e.g. lion, leopard and spotted hyaena were completely absent in cultivated areas. This conversion of habitat also affects large carnivores indirectly by reducing abundance of prey particularly large prey which are an important component in the diet of large predators. Large predators need large prey in order to maximise food intake and minimise energy expenditure for hunting (Carbone and Gittleman 2002). In this study I found lower relative abundance of large prey outside the park particularly in the cultivated areas.

Hunting outside the Tarangire National Park may also be another reason for the observed low abundance of large carnivores in pastoral grazing areas and cultivated areas. Hunting is carried out in two ways: Licensed wildlife hunting through the Wildlife Division which allows hunting of carnivores for trophies particularly large carnivores e.g. lion and leopard and illegal hunting. Hunting may affect carnivore abundance directly through removal of individuals in the population or indirectly through depletion of prey base which may be both through licensed hunting and poaching. Illegal hunting in this area is mainly for meat from ungulates and its impact may be particularly important for large carnivores because large ungulates are the ones that are potentially hunted by poachers as they are more profitable. Throughout my stay in Loiborsoit village during the field work, I witnessed a substantial number of illegal hunters arrested by park rangers with zebra carcases. Furthermore hunting may also affect species behaviour which may in turn affect species capture success and therefore relative abundance.

As the intensity of human activities increases not only large carnivores are affected but also medium-sized and less wide ranging species which are considered to be common also tend to decline. For example, black-backed jackal is generally common in the Tarangire ecosystem and across their range in Africa (Sillero Zubiri et al. 2004) and the species was found in all land use types in the ecosystem. However relative abundance of this species was lower in the cultivated areas compared to the park and grazing areas which suggest that environmental change can have significant impact even on species that are considered to be common depending on the intensity of disturbance. However it is worth also noting that in addition to human activities, the observed variation in relative abundances of species between the three land use types partly may be because differences in trapping effort e.g. changes in the trapping regime from 2 km camera spacing to 1 km in part of the grid in the park and loss of cameras outside the park in the pastoral grazing areas and cultivated areas may influence capture success. But also differences in relative abundances between land use types may also be because of ecological differences which naturally allow the park to support high abundance of most wildlife species. For example, the presence of permanent water during dry season in the Tarangire River and swamps such as Silale and diversity of habitat types - the microphyll and decidous savannah habitats found in the park are thought to be important for maintaining populations of most wildlife species in the ecosystem (Lamprey 1963, Lamprey 1964, Van De Vijver et al. 1999).

Overall these results show that despite no statistical significant difference in carnivore species richness between the park and pastoral grazing areas that were discussed in chapter 3, relative abundance of carnivores is lower outside the park in pastoral grazing areas and cultivated areas. This shows that while species richness may be higher, species abundance may be lower due to human activities. However as discussed above it is also possible that other factors that were not investigated here may also be important e.g. the park was established in an area that naturally support higher abundance of wildlife due to diversity of habitats (Lamprey 1963, Lamprey 1964, Van De Vijver et al. 1999). Variation in habitats and physical features such as topography may influence species detections and therefore relative abundance (O'Connell et al. 2006).

4.5.1.2 Non-carnivores

These analyses have also shown that photographic rates and the number of cameras that photographed non-carnivores and the probability of detecting non-carnivores was higher in the park than in the pastoral grazing areas and cultivated areas. These differences in non-carnivore relative abundances can also be explained by variations in human activities and the extent of use between the park and grazing areas and cultivated areas as well as other factors that were discussed above. Nonetheless previous survey in the ecosystem (TWCM 2000) showed also lower abundance of wildlife populations outside the park which is also attributed to human activities, especially the expansion of agriculture and illegal wildlife hunting. Cultivation in the Tarangire ecosystem has increased tremendously over the last three decades due to expanding human populations.

Variations in relative abundance were also observed between large and medium-sized and small species. Generally, large non-carnivores were less abundant especially outside the park, which also suggests they are more sensitive to disturbance than smaller and medium-sized species. As discussed in chapter 3, this is because large species need extensive habitats to survive, yet such habitats are difficult to maintain because of the increasing human pressure due to expanding human population and demand for land and other resources outside protected areas. But also generally large mammals are potentially the target of most poachers because they are most profitable. In Ivory Coast it was found that populations of large ungulates declined in the Comoé National Park because of intensive illegal hunting large ungulates (Fischer and Linsenmair 2001). Consequently the conservation of large non-carnivore prey depends on the existence of a network of protected areas with effective management. Previous analyses based on aerial census of large herbivores in Tanzania (Stoner et al. 2007) also found higher abundance of large species in strictly protected area such as national parks than in other forms of protected and non-protected areas thus demonstrating the importance of strictly protected areas in maintaining mammal populations.

Variations in relative abundance of non-carnivores were also expected due to temporal variations in species detections due to seasonal migration exhibited by some species in the Tarangire ecosystem (Kahurananga and Silkiluwasha 1997, Gereta et al. 2004), see also Voeten (1999), and was the case for this study. Differences in species relative abundance between land use types was significantly higher for species with marked seasonal movements e.g. zebra were not photographed during the wet season in the park but were photographed during the dry season. This is because the species migrates outside the park during wet season and returns during the dry season (Kahurananga and Silkiluwasha 1997, Gereta et al. 2004). Surprisingly variations in prey relative abundances were also observed in territorial species such as hartebeest which suggests that local movements were also important during this study (see chapter 3 for details). But it is also possible that the species were present during the survey but were not detected by cameras which may be due variation in habitat conditions between wet and dry season such as vegetation cover. For instance, Tarangire National Park is characterised by tall grass during the wet season than during the dry season which may affect detectability of species.

4.5.2 Occupancy and Habitat Use

Effect of land use type on species occupancy or habitat use showed that most species had higher occupancy and habitat use in the park than in the pastoral grazing areas, which suggest that the species were relatively more abundant in the park than in the pastoral grazing areas.

4.5.2.1 Carnivores

Two species (banded mongoose and black-backed jackal) were the only ones which showed that land use type had a significant effect on the species occupancy. The reason that fewer carnivore species than non-carnivores showed significant results may be that fewer cameras detected carnivores. Carnivores are difficult to detect because they are cryptic but also some species occur at low densities and therefore the number of cameras that detect species is likely to be lower. However it is possible that with increased trapping effort occupancy of most species could have been estimated. The lower occupancy of banded mongoose and black-backed jackal in pastoral grazing areas may also be explained by the variations in human activities and extent of human use between the park and ecological differences between pastoral grazing areas and the park as discussed in previous sections. But also this lower occupancy of black-backed jackal in pastoral area suggests that carnivores are more sensitive to human activities even for wide spread species such as black-backed jackal and banded mongoose (Kingdon 1997, Sillero Zubiri et al. 2004). Overall it can be said that differences in carnivore occupancy between the park and pastoral grazing areas may be due to combined effect of factors such as human activities, sampling effort as well as ecological difference between the two land use types.

4.5.2.2 Non-carnivores

Non-carnivore results show occupancy and habitat use were higher in the park than in the grazing areas which suggests that non-carnivores were more abundant in the park than in the grazing areas. As discussed in previous sections on carnivores above, this variation in non-carnivore species occupancy and habitat use may be influenced by differences in human activities between land use types, differences in trapping effort and ecological factors. The higher number of non-carnivore species that showed significant effect of land use on species occupancy may be because non-carnivore occupy lower position in the trophic level and are likely to be more abundant than carnivores.

From the discussion above it can be said that the lower relative abundances of mammals outside the park found here may be because of combination of factors. On one hand anthropogenic activities such as land use change appear to be important although on the other hand it is also possible that other historical factors such as the park may have been designated in an area naturally rich in species cannot be completely discounted. Difference in the trapping effort may also have contributed to the observed differences in species relative abundances between land use types. A similar study in future will be important in order to understand the importance of these factors in influencing relative abundance of species in the area.

4.5.3 Implications for Wildlife Conservation

The study has shown two important results which are important for wildlife conservation. First, species relative abundances decreased from the park to the grazing areas and cultivated areas which suggest that protected areas are important for maintaining wildlife populations, probably because they exclude human activities, but it is also possible that other factors such as ecological differences between land use types might be important. Second, these results have shown a potential for monitoring a range of species using relative abundance from camera traps. However there are also limitations associated with the use of camera traps. These limitations will be discussed in chapter 7. Although the use of relative abundance to monitor species provides valuable insights about species habitat use and responses to environmental change, often conservation planning requires specific knowledge of species population sizes, for example for estimating longer term population viability of threatened species and managing species which have negative impact on the environment or people’s livelihoods. Information on species absolute abundances is extremely important for conservation planning. This topic is covered in detail in chapter 5.

CHAPTER 5: ESTIMATING DENSITY OF INDIVIDUALLY RECOGNISABLE SPECIES IN THE TARANGIRE NATIONAL PARK

5.1 Summary

The aim of this chapter was to determine the density of individually recognisable carnivores in the Tarangire National Park and to explore the potential for monitoring species from camera trap photographs. Remote cameras were used to take photographs of animals in the park. I was able to estimate absolute density of three species out of the seven individually recognisable species which were photographed in the park. These species were leopard, serval and aardwolf. Leopard density was estimated to reach 7.9 ± 2.09 animals per 100 km^2 during the wet season. It was not possible to estimate leopard density in the dry season due to very few captures and no recaptures at different camera stations. Serval density was estimated at 10.9 ± 3.17 animals per 100 km^2 during the dry season while no density estimates were available during the wet season due to lack of recaptures. Aardwolf density was 9.04 ±2.54 animals per 100 km^2 during the dry season but no density estimates were determined during the wet season due to a lack of recaptures. No density estimates could be determined for spotted hyaena, common genet, large spotted genet, and wild cat due to a combination of low capture rates and lack of recaptures at different camera stations. Results suggest that the spatial-temporal variation in prey abundance and the distance between cameras could influence species capture success. The use of camera-trap data in combination with capture-recapture models has good potential of monitoring populations of species that can be individually identified in protected areas, which are not easily monitored by other means.

5.2 Introduction

Management of wildlife populations sometimes requires information of species absolute abundances (Stander 1998, Hopkins and Kennedy 2004, Sutherland 2006). This is especially important for threatened species because such information is needed for determining the viability of the population (Reed et al. 2002, Linkie et al. 2006). Absolute abundance is also useful for setting sustainable hunting quotas in areas where species are hunted (Baldus and Cauldwell 2005, Lindsey et al. 2007, Lindsey 2008), and is also potentially important for determining the number of individuals that a reserve can support (Dillon and Kelly 2007). Information on abundance may also be required by wildlife managers in order to meet some specific management objectives such as population control of certain species such as elephants that may cause environmental damage or may negatively affect people's livelihoods (Van Aarde et al. 1999, Morellet et al. 2007) or for species that play significant role in the functions of ecosystem (keystone species) e.g. wildbeest in the Serengeti (Mduma et al. 1999).

Direct methods such as aerial census (Seber 1992) and distance-based sampling methods (Buckland et al. 1993) are widely used to census animals. However these methods are most suitable for large herbivores and in open areas where animals can be easily seen, so may not be suitable for low density and cryptic species, such as most carnivores (Smallwood 1993, Stander 1998). Over the last three decades camera traps have been frequently used in combination with capture-recapture models to estimate density of mammal species that can be individually recognised. The method has been used to monitor a wide range of carnivores, e.g. tiger (Karanth and Nichols 1998), jaguar (Maffei et al. 2004, Silver et al. 2004, Di Bitetti et al. 2006), ocelot (Trolle and Kery 2005, Dillon and Kelly 2007), puma (Kelly et al. 2008) and leopard (Henschel et al. submitted). However new camera trap techniques and analytical methods are also being developed. For example Rowcliffe et al., (2008) have developed a camera trap-based technique for estimating density of animals based on their specific daily range and daily activity patterns, a technique which does not require identification of individual animals. However at the moment the method has only been tested with captive animals where daily ranges can be easily determined (Rowcliffe et al. 2008), wider application of this technique in the wild depends on better information on animal daily ranges.

As discussed in chapter 2, the use of camera traps to estimate the density of mammals has largely been conducted in tropical forest habitats, such as Argentina (Di Bitetti et al., 2006), Belize (Dillon and Kelly 2007) and India (Karanth and Nichols 1998). The potential of camera traps as a tool for monitoring mammals in the African savannas has not been investigated. This is because camera traps have only been introduced over the last decade in Africa when their use was focussed on forest habitats. Furthermore, most of the previous studies have focused on single species, whereas it is likely to be more economical to estimate the density of several species simultaneously from a single study. Therefore, a wider understanding of the potential of camera traps for monitoring carnivores is important. In particular, as carnivore species vary in home range size (Gittleman and Harvey 1982), an understanding of species' home ranges can be used to determine appropriate camera spacing to obtain sufficient recaptures for density estimates (Karanth and Nichols 2002, Silver et al. 2004).

The overall aim of this chapter is to use camera traps to estimate the density of a wider carnivore community, to determine the density of individually recognisable carnivores in the Tarangire National Park, and specifically to answer the three following questions:

1. What is the density of individually recognisable carnivores in the Tarangire National Park?

2. Does camera spacing influence species capture success?
3. Are camera traps potentially useful for determining densities of carnivores in the region?

5.3 Material and Methods

A more detailed background to camera trapping methodology was presented in chapter 2. In this chapter I will describe the application of camera traps for estimating absolute abundance of individually recognisable species. Surveys for estimating species abundance were carried out in the Tarangire National Park only. The survey design was based on a previous study which showed the potential for using camera traps in the Tarangire National Park to estimate absolute abundance of leopard (Kelly et al., unpublished data).

5.3.1 Survey Design

The use of camera traps to estimate species abundance is based on capture-recapture models (White et al. 1982). The models have two important assumptions which must be satisfied when designing a camera trap survey. The first one assumes that no animal within the population has a zero chance of being captured or photographed. This assumption is important because it dictates how far apart the cameras should be placed while ensuring that there are no gaps between camera stations which fall entirely within an animal's home range. In order to satisfy this assumption, a traditional approach has been to adopt the smallest known home range of a species in the study area, as the minimum area within which there must be at least one camera station (Karanth and Nichols 2002, Silver et al. 2004). For this study serval was the smallest species in the study area whose home range has been documented, with areas 9.5 and 11.6 km^2 for adult female and male serval respectively (Geertsema 1985). Furthermore, serval was also one of the target species for estimating absolute abundance. Therefore, based on the limited published information on serval home ranges, in the first survey cameras were placed at 2 km intervals.

The second assumption for capture-recapture models which needs to be satisfied is that the population is closed throughout the survey i.e. there are no births, deaths, migration or emigration (White et al. 1982). This assumption is important because it dictates how long the camera survey should be. In most camera trapping studies population closure can be approximated based on life history characteristics of the species by ensuring that the camera trapping period is kept short (Karanth and Nichols 1998), In order to satisfy the closed population assumption, cameras for this study were left to operate 24 hours a day for a maximum of 75 days. This period was chosen as a compromise between attempting to ensure that the population closure assumption was not violated, but also attempting to obtain a minimum of 1000 camera trap days, a level which is required for a robust statistical analysis (Carbone et al. 2001). For long lived species such as carnivores it is unlikely that there will be large number of deaths and births within this period to violate the closed population assumption (White et al. 1982).

5.3.2 Data Collection

In order to estimate absolute abundance the survey targeted species in the Tarangire National Park that could be individually identified from spot patterns or stripes. The protocol used here was as follows: Two surveys were conducted in the national park, the first in 2006 and the second in 2007. In 2006, the survey protocol was as described in chapter 4 i.e. 20 camera stations with two cameras at each station during the wet season (March – May) with a spacing of 2 km between stations, covering an area of 48 km^2. But in 2007, the survey was conducted during the dry season (June - August). This later survey also used a 48 km^2 grid of camera traps but with 42 camera stations. Two cameras were again placed at each station but the spacing between stations was reduced to 1 km (Figure 13). The increase in the number of camera traps from 20 to 42 and the reduction of camera spacing from 2 km to 1km was designed in order to increase the likelihood of recapturing individuals at different camera stations, to enable estimation of density for species with smaller home ranges (Karanth and Nichols 1998). This was because preliminary results from 2006 survey showed that most target species (serval, common genet, aardwold and spotted hyaena) were not recaptured at different camera stations, which was necessary for the use of the mark-recapture framework. In both surveys cameras were set at approximately 40 cm above the ground and all cameras were checked regularly to replace batteries and films. Films were developed and printed and all mammals could be identified to species level. Spot paterns and stripes were used to indentify individuals for subsequent analysis.

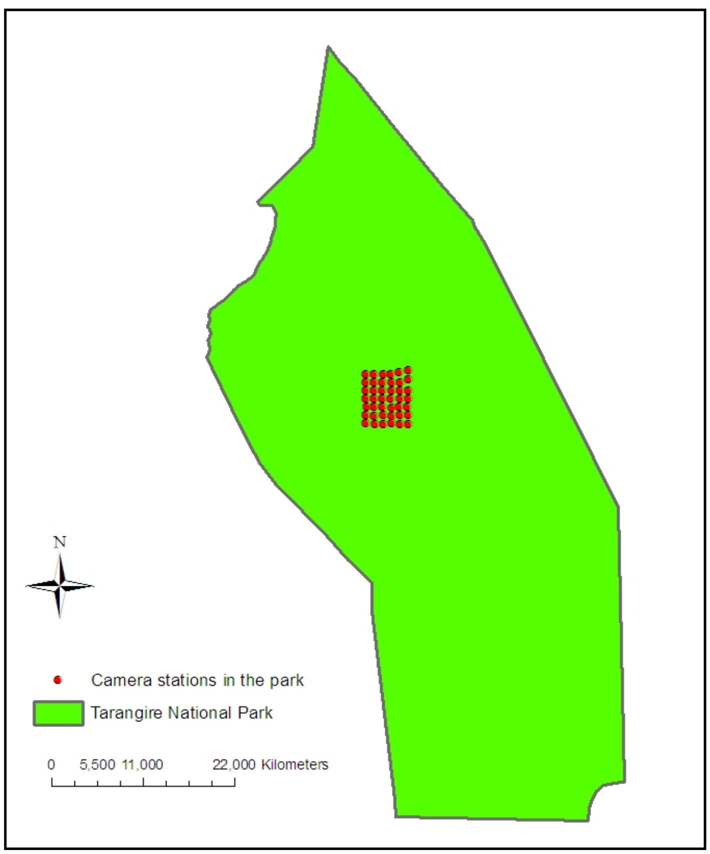

Figure 13: Location of camera traps in the Tarangire National Park for 1 and 2 km combined.

5.3.3 Data Analysis

Within my study area there were five species photographed with potential for estimating absolute abundance through mark-recatpure methods (leopard, aardwolf, serval, spotted hyaena and common genet). Individuals of these species could be identified using spot patterns and body stripes. For individually recognizable animals which could undergo multiple captures, a capture history could be compiled. The capture history of an individual consisted of value of ‘1’ if the animal was photographed in a given occasion or ‘0’ if the animal was absent. However if there are many zeros in the capture matrix, it is unlikely that the program CAPTURE will work well. In order to address this the survey period (75 days) was broken into intervals of 10 days – trapping occasions. Thus there were a total of eight trapping occasions over the entire survey period. A capture history matrix of all individually identified animals was then compiled for each species: the matrix consisted of columns representing individuals and rows representing trapping occasions. The matrix was then used in the program CAPTURE (White et al. 1982, Rexstad and Burnham 1991) to estimate abundance of the species concerned. The program allows seven different models to be used in order to generate abundance estimates for the sampled area, and the choice of model is based on the species biology, as well as being dependent on the number of individuals that have been captured and the frequency of recaptures (Otis et al. 1978, Rexstad and Burnham 1991). These models differ in their assumed sources of variation shaping the recapture probabilities of individuals. The sources of variation considered in the CAPTURE models are individual heterogeneity e.g based on sex, age, ranging patterns and behavioural response and time (Otis et al. 1978, White et al. 1982, Rexstad and Burnham 1991). Individual heterogeneity refers to variation in capture probability among individuals, such that an animal is thought to have its own capture probability which may differ from the capture probabilities of the other individuals, while behavioural response refers to the changes in capture probability that occurs after an individual is caught for the first time. This means that at any sampling occasion uncaptured and previously captured animals may have a different capture probablity. Time variation simply refers to variation in capture probability from one sampling occasion to another. The program CAPTURE computes a closure statistic to test the closed population assumption for each data set (Otis et al. 1978, White et al. 1982, Rexstad and Burnham 1991).

The number of animals computed by the program CAPTURE was divided by the survey area to obtain density which is expressed as the number of animals per 100 km^2. However, because it is not possible to achieve complete spatial closure because an animal’s home range may go beyond the trapping grid (White et al., 1982), the estimation of density involved determining a buffer that can be added to the grid to obtain the effective sampling area. I followed Silver et al, (2004a) to estimate the mean maximum distance moved among multiple captures (MMDM) and applied half (1/2MMDM) as a buffer to each camera station to obtain effective survey area. The effective survey area was then used for estimating density by dividing the number of animals computed by the program CAPTURE by the effective survey area.

5.4 Results

5.4.1 Species Abundance

The number of photographs and individuals identified for each species varied considerably with season. Generally, more photographs were taken and more individuals identified during the wet season than in the dry season. Similarly, overall capture success was higher in the wet season than in the dry season (Table 10).

5.4.1.1 Leopard

A total of 17 leopard photographs were obtained in this study from 3758 camera trap days. The number of leopard photographs recorded and individuals that were identified varied significantly between the wet and dry season. More photographs were obtained during the wet season (14) than during the dry season (3). Similarly the number of individuals identified were higher during the wet season (4) than during the dry season (2) (Table 10). Investigation into leopard movements within the grid revealed that during the wet season two leopards were recaptured at different camera stations while during the dry season only one individual was recaptured at a different camera station. The increase in the number of camera traps from 20 to 42 in the grid and the reduction in camera spacing from 2km to 1 km did not increase the number of recaptures during the dry season. Using the program CAPTURE and capture-mark-recapture statistical models, leopard capture probability during the wet season was estimated at p = 0.18 (Table 11) leading to an abundance of 5±1.32 individuals. The Jackknife heterogeneity model (Mh) was selected as the best fit model (Table 12) and showed that that each leopard had a unique capture probability. A closure test indicated that there was no violation of the closed population assumption (Table 12). The total distance moved by the two individual leopards was 4 km and produced a buffer of 1 km. Leopard density for the wet season was estimated at 7.9 ± 2.09 animals per 100 km^2 using effective sample area of 63 km^2 (Table 12). For the dry season only two individuals were identified from three photographs and there were no recaptures at different camera stations, so no density estimate could be determined.

5.4.1.2 Serval

A total of 26 photographs of serval from a total of 3758 camera trap nights were obtained from the two surveys. More photographs were recorded during the wet season (14) than during the dry season (12). Similarly the number of individual servals identified was higher during the wet season (6) than during the dry season (4). No individuals were recaptured at different camera stations during the wet season but two individuals were recaptured during the dry season and therefore density estimates were determined for the dry season only.

Results showed that the capture probability for serval during the dry season reached $p = 0.20$ and the abundance estimate was 6 ±1.75 individuals. The Jackknife (Mh) model was again selected as the best fit model which shows that each serval has a unique capture probability. A closure test indicated that there was no violation of the closed population assumption (Table 12). The total maximum distance moved by two individual servals was 2 km and produced a buffer of 0.5 km: serval density was thus estimated as 10.9 ± 3.17 animals per 100 km^2 using an effective sample area of 55.25 km^2 (Table 12).

5.4.1.3 Aardwolf

A total of 36 photographs of aardwolf were obtained from a total of 3758 camera trap nights in this study. More photographs of the species were recorded during the wet season (29) than during the dry season (7). Similarly the number of individual aardwolf photographed was higher during the wet season (29) than during the dry season (7). There were no recaptures of aardwolf during the wet season and only one individual was recaptured during the dry season. Results from the capture-recapture analysis showed that the capture probability for aardwolf during the dry season was high ($p = 0.26$), and the abundance estimate was 5 ±1.60 individuals. The Jackknife null model (Mo) was selected as the best fit model and showed that there was no variation in capture probability of aardwolf due to time, behaviour, or individual heterogeneity. A closure test indicated that there was no violation of the closed population assumption (Table 12). The maximum distance moved was 1 km by one individual aardwolf giving a buffer of 0.5 km. Thus density estimates for aardwolf was 9.04 ± 2.54 animals per 100 km^2 for a 0.5 km and area of 55.25 km^2.

5.4.1.4 Spotted hyaena

A total of 25 photographs of spotted hyaena were recorded from a total of 3758 camera trap nights in this study. More photographs of the species were recorded during the wet season (18) than during the dry season (7). Similarly the number of individual spotted hyaena identified was higher during the wet season (13) when 20 camera stations with 2km spacing were used than during the dry season (3) when 42 camera stations with 1km spacing were used. However, in both surveys no recaptures were obtained and therefore no abundance estimates could be determined.

5.4.1.5 Common genet

The common genet was the most abundant individually identifiable species in the Tarangire National Park. A total of 66 photographs of the species were obtained from 3758 camera trap nights. The number of photographs obtained varied with season, and more photographs were obtained during the wet season (42) than during the dry season (24). Similarly the number of individual common genets identified was higher during the wet season (29) than during the dry season (12). Nonetheless there were no recaptures in either seasons and therefore no density estimate could be determined.

Table 10: Summary of the number of species photographs, recaptures, number of individuals and unknown individuals for wet and dry season surveys in the park, and the number of photo-captures per 100 trap days (camera trap success).

Species	No. photos		Recaptures		Individuals		Unknown		Captures/ 100 trap days	
	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry
Leopard	14	3	9	3	4	2	0	1	0.14	0.03
Serval	14	12	0	6	6	4	0	0	0.14	0.12
Aardwolf	29	7	0	6	11	4	4	0	0.29	0.07
Spotted hyaena	18	7	0	0	13	3	2	0	0.18	0.07
Common genet	42	24	0	0	29	12	4	0	0.42	0.24

Table 11: Species captures in the park with estimated capture probability (p) under the Jackknife heterogeneity model M(h) and null model M(o).

Species	No. individuals	No. captures	Male	Female	Unknown sex	Capture probability (*p*)
Leopard	4	9	2	1	1	0.18
Serval	4	6	-	-	12	0.20
Aardwolf	4	6	-	-	7	0.26

Table 12: Results of closure test for the closed population assumption and the abundance estimates of leopard, serval and aardwolf in the park using the Jackknife heterogeneity model M(h) and null model M (o).

Species	Abundance ±SE	95% CI	Closure test z	Closure test p	Buffer 1/2MMDM(km)	Density/ $100km^2$ (SE)
Leopard	5 ±1.32	4 – 12	-0.83	0.20	1	7.9 ± 2.09
Serval	6 ± 1.75	3 – 13	0.70	0.76	0.5	10.9 ± 3.17
Aardwolf	5 ± 1.60	5 – 14	-1.29	0.10	0.5	9.0 ± 2.54

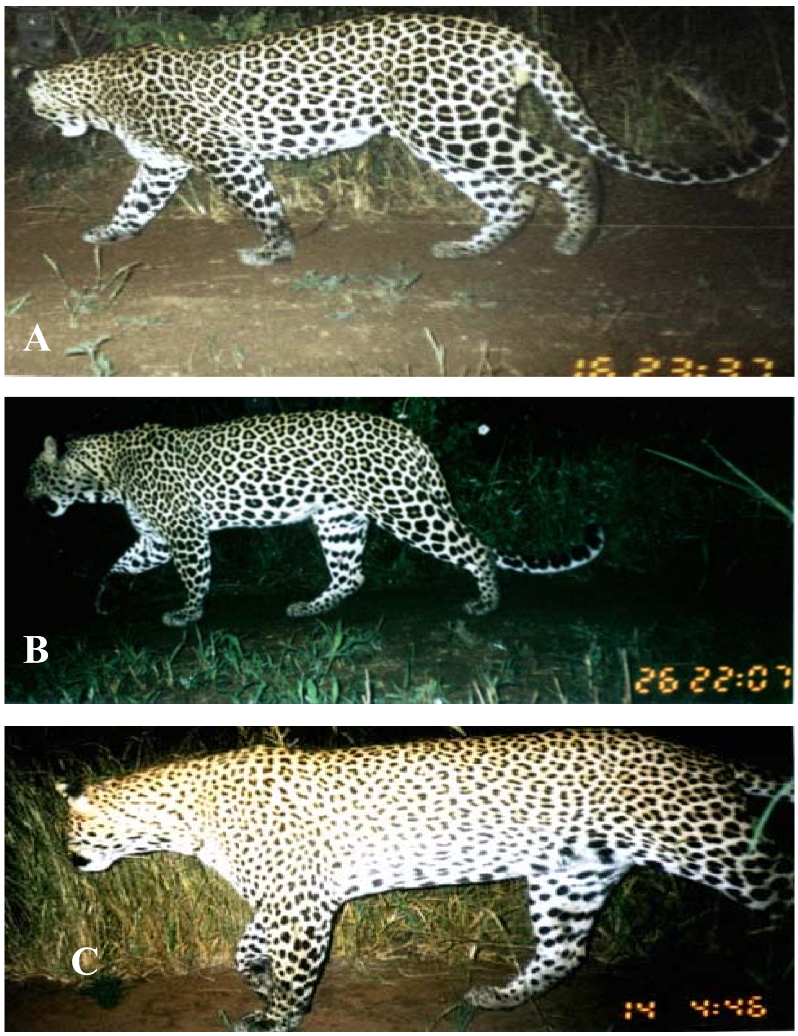

Plate 4: Two individual leopards caught by cameras. A and B show male leopards (T18). The spot patterns on A and B are indeed identical in shape and location on the animal. C shows a different animal (T16).

5.0 Discussion

This study has shown that photographic capture-recapture studies provide a potentially useful approach for estimating density of individually identifiable species. These results present one of the first studies to use the technique for estimating density of a carnivore community. Density estimates were determined for three individually identifiable species, leopard, serval and aardwolf out of seven species that could have potentially been estimated in Tarangire National Park. This suggests that camera traps, combined with capture-recapture techniques, could be used to simultaneously monitor more than one species of individually identifiable carnivores. The density of leopards was 7.9 ± 2.09 animals per 100 km^2 while that of serval was 10.9 ± 3.17 animals per 100 km^2 and 9.4 ± 2.54 animals per 100 km^2. There were no recaptures for common genet and spotted hyaena, and hence model convergence was not possible for these species.

5.1 Leopard

These results show that leopard captures were very variable between seasons. Capture success was higher during the wet season when 14 photographs were obtained, from which four individuals were identified, giving a density estimate of 7.9 ± 2.09 animals per 100 km^2. However, during the dry season only three photographs were obtained and two individuals identified, and hence no density estimates could be determined due to lack of recaptures. It was surprising that despite having higher capture rates during the wet season, the density estimates obtained appears to be relatively low, yet leopards are generally thought to be common in the Tarangire National Park (Foley 2004) and they are also known to be territorial and are therefore would be expected to be present throughout the year (Bailey 1993). Furthermore camera spacing was reduced from 2 km during the wet season to 1 km during the dry season and the number of camera stations were increased during the dry season in an attempt to increase recaptures at different camera stations.

There are three possible explanations to the observed variation in leopard capture rates. One possibility is that, this variation may be due to variation in prey abundance and distribution between the wet and the dry season. Although leopards feed on a wide range of species (Henschel et al. 2005, Hayward et al. 2006), the abundance and distribution of their prey is one of the factors that affects the species abundance in many areas (Henschel et al. submitted) as well as other large carnivores. In the Tarangire National Park, most herbivores, except migratory species, are more evenly distributed during the wet season because pasture and water are plentiful everywhere, but during the dry season prey distribution would be expected to be more clumped, especially along the Tarangire River basin, where animals search for green pasture and water and studies elsewhere have shown that availability of water and pasture influence the abundance and distribution of herbivores in the savannahs (Thrash et al. 1993, 1995). Optimal foraging theory predicts that a consumer should maximise food intake and minimise energy expenditure in obtaining food (Dean and Siegfried 1991), and hence leopards would be expected to move to areas with abundant prey within their home ranges in the study area, in order to maximise food intake. The camera trap grid covered part of the Tarangire River basin, although not the entire basin.

Another possibility is that, the observed variation in leopard captures may also be due to variation in the availability of travel routes between dry and wet season. During the wet season, grass is often very tall in the park and therefore travel routes may be fewer, compelling most individuals to use the few available routes. However, during the dry season the habitat is more open and trampling by mammals is very high as migratory species return to the park. The higher concentrations of wildlife in the park during the dry season creates more routes toward areas with green pasture and water and these routes may provide more opportunities for leopards and other species to choose which trail to use and therefore lead to lower capture rates. In a camera trap study of leopards in tropical forest in Cameroon, Henschel and Ray (2003) found that a major factor affecting leopard capture probability was the availability of travel routes. They showed that leopard capture rates were higher in areas with fewer trails than in areas with dense trails.

It is also suggested that when species capture rate is low its may be an indication that the species density is too low to achieve estimates of density regardless of the effort employed (Karanth and Sunquist 2000). However in the Tarangire National Park, leopards are generally thought to be common although there are no published density estimates (Foley 2004). It is therefore likely that variation in prey abundance and travel routes influenced leopard capture in the study area.

Previous attempts to estimate leopard density in protected areas in Tanzania and elsewhere in Africa have mainly been based on animal body mass (East 1984). Leopard density estimates from camera trap studies are few e.g. by Henschel et al., (in press), mainly because the use of camera traps to study mammals has only been introduced recently in Africa. However, the estimates from other studies are comparable to the results of this study. Reported leopard densities using extrapolations from prey biomass include 5.6 animals per 100 km^2 in the Serengeti (Schaller 1972), 7.5 animals per 100 km^2 in the Ngorongoro and 11 animals per 100 km^2 in Lake Manyara National Park (East 1984). Estimates based on biomass from other conservation areas in Africa include Queen Elizabeth National Park, 16 animals per 100 km^2 (Van Orsdal 1981), Nairobi National Park 8.5 animals per 100 km^2 (Rudnai 1973), Kruger National Park 2.5 animals per 100 km^2

(Mills and Biggs 1993), Kaudom Game Reserve 1.5 animals per 100 km^2 (Stander et al. 1997) and Hwangwe National Park 2.1 animals per 100 km^2 (Wilson 1975). The only published information using camera traps are from west Africa, where leopard density estimates from Ivindo National Park study areas were 4.58 ± 2.58 animals per 100 km^2 in Dilo and 12 .08 ± 5.11 animals per 100 km^2 in Massouna (Henschel et al. submitted). It is suggested that these differences in leopard densities in sub-Saharan Africa are due to variation in environmental conditions, particularly rainfall and soil nutrients which affect prey biomass (East 1984). However most of these conservation areas listed above do not have similar habitats to that found in the Tarangire National Park - the Southern acacia-commiphora bushlands and thickets ecoregion (www.worldwilfe.org/science).

5.2 Serval

Despite servals being widespread across its range in Africa information on the species density is very limited. To my knowledge, the only comprehensive study of serval comes from the Ngorongoro Conservation Area by Geertsema (1985), who estimated density of serval in the Ngorongoro Crater at 1 animal per 2.4 km^2 from 33 known individuals or 41.7 animals per 100 km^2. However, not all the entire crater was surveyed and therefore this estimate is thought to be under estimate (Geertsema 1985). The results from our study provide a valuable contribution to understanding of the ecology of serval. Capture success for serval in the Tarangire National Park varied between seasons. Capture success was higher during the wet season when 14 photographs were obtained from which 6 individuals were identified, but there were no recaptures at different camera stations and therefore density could not be determined. But during the dry season 12 photographs were obtained from which four individuals were identified that gave a density of 10.9 ± 3.17 animals per 100 km^2. This difference in capture rates between the wet and dry season may be due to the difference in availability travel routes as discussed earlier.

Comparison of density estimates obtained from this study and that of Ngorongoro Crater by Geertsema (1985) shows that the Tarangire estimates are lower. This difference may be due to differences in habitats between the study areas. For instance, in Tarangire the surveyed area consisted mainly of wooded savannah and a small section of the Tarangire River basin which has tall grass, while in the Ngorongoro Crater, the survey was carried out in medium to tall grasslands surrounded by swamps. The latter are preferred habitats for serval and are important for prey species such as rodents and birds and therefore may lead to higher densities (Geertsema 1985, Estes et al, 1991, Kingdon 1997). Interestingly, when compared to ocelot, a species similar in size to serval, the density of ocelot appear to be lower in some study areas and higher for others, depending on type of habitat e.g. Dillon (2005) estimated the density of ocelot in rainforest to range from 18.91 to 20.75 ocelots per 100 km^2 while in pine forest density of ocelot ranged from 2.31 to 3.80 ocelots per 100 km^2 compared to 10.9 ± 3.17 animals per 100 km^2 for serval that was obtained here.

5.3 Aardwolf

The ecology of the East African subspecies of aardwolf is poorly known, as most studies have been focussed on the southern Africa subspecies (van Jaarsveld et al. 1995, Sliwa 1996, Taylor and Skinner 2000), see also (Fagerudd 2005). This study provides the first density estimates of the aardwolf in Tarangire National Park. Results showed significant differences in photographic captures of the species between the wet season when 29 photographs were obtained and the dry season when only 7 photographs were obtained. Despite the higher number of captures during the wet season, no individual was recaptured at different camera stations and therefore no density estimate could be determined, although four individuals could be identified. During the dry season, recaptures were obtained and density was estimated at 9.04 ± 2.54 animals per 100 km^2. The lack of recaptures at different camera stations during the wet season was surprising because aardwolf are known to be territorial (Richardson 1987a) and the two and half month survey period was thought to be sufficient to capture the species without violating the closed population assumption. The lack of recapture at different camera stations during the wet season may be due to the camera spacing used here. The 2 km spacing used during the wet season was probably too big for aardwolf because the species home range is relatively small (1-3km^2) (Richardson 1987a). The reduction of camera spacing from 2 km to 1 km had a positive effect, although still only a single individual was recaptured at different camera stations.

The lower capture of aardwolf obtained during the dry season may be also be due to variation in prey availability and foraging behaviour of the aardwolf. Aardwolves are known to be solitary and nocturnal and are specialised in their diet. The species feed mainly on two types of harvester termites *Trinervitermes* and *Hodortermes* (Richardson 1987b, Kingdon 1997, Williams et al. 1997). Studies have shown that the harvester termite availability is affected by temperature i.e. at higher temperatures the termites remain inside the termite mounds and only come out at favourable temperatures and humidity (Seely and Heinrich 1981, Richardson 1987b). In the Tarangire National Park dry season is characterised by high temperatures and low humidity during the day, whilst temperatures can be low during the night, whereas during the wet season temperatures are more moderate and humidity is higher, which is likely to make termites more active and may lead to increased aardwolf activity. During the field survey in the wet season, termite mounds appeared to be very active compared to the dry season. In some areas studies have also shown that aardwolf may associate with feeding aardvarks because aardvarks are adapted to opening termite mounds, whereas aardwolf feed on termites exposed on the surface by the aardvark (Taylor and Skinner 2000). The number of cameras that photographed aardvark during the dry season was higher compared to wet season (chapter 4) which may suggest that lower capture of aardwolf during was probably not influenced by aardvark.

It is also possible that variation in trail density between wet and dry season might have had an influence in capture rates as in discussed in previous sections. For instance, despite reducing camera spacing from 2 km to 1 km during the dry season, only one individual was recaptured at different camera stations. It was expected more individuals would have been captured this time given that they have relatively a smaller home range (1-3km^2) (Richardson 1987a). Higher trail densities during the dry season might have provided opportunities for aardwolf to use different trails, and avoiding the camera traps. Overall it can be concluded that, given that aardwolves have been shown to be territorial (Richardson 1987a, Kingdon 1997), camera spacing and variation in trail density and prey availability between wet and dry seasons may influence the species capture. It is also possible that other factors might also be responsible e.g. the species may be present but not be detected and so a future study on the ecology of this species in Tarangire ecosystem is important in order to design an appropriate monitoring survey.

5.4 Influence of Camera Spacing and Spatial Coverage on Species Captures

Overall these results have demonstrated that a spacing of 1 km was required for estimating densities of serval and aardwolf whilst 2 km was more appropriate for leopard. This variation in camera spacing was important because home ranges of carnivores vary in size. Smaller species usually have small home ranges compared to larger species (Gittleman and Harvey 1982), and these differences pose a challenge when designing surveys for multiple species. There is a trade off between the need for increasing spatial coverage in order to ensure that home ranges of large carnivores are included in the trapping grid, and the need for reducing camera spacing in order to recapture smaller species. Dillon and Kelly (2007) have shown that it is difficult to strike a balance between camera spacing and the need to cover large areas for species with large home ranges without compromising the capture of smaller species. However this problem can be partially addressed by increasing the survey effort in terms of number of cameras, so that a large area and small camera spacing can be combined. But it is important also to understand that this has cost and feasibility implications, because it requires deployment of many more camera traps and it is not always feasible to put up a large number of traps in a short period of time.

5.5 The Potential of Camera Traps for Estimating Densities of Carnivores

The use of camera traps to estimate carnivore densities has largely been in forested habitats where animals may be forced to use established routes. This study is one of the few to be conducted in savannah habitats and therefore provides the opportunity to test the performance of the method in savannahs. Results showed that camera traps have the potential for estimating densities of carnivores in the Tarangire ecosystem in combination with capture-recapture techniques. Density estimates were determined for about half of the individually recognisable species in the park which suggest that the use of camera traps has potential for monitoring these species. Refinement of the survey design might lead to further improvements. The problem of low recaptures found here may be partially addressed by identifying a single target species for which a specific camera trap monitoring protocol can be developed or by increasing survey effort by deploying more cameras.

Whilst the determination of the abundance of wildlife populations is important in the development and implementation of effective conservation strategies, the long-term survival of wildlife ultimately depends on the ability of people adjacent to wildlife areas to coexist with wildlife. This coexistence depends largely on the attitude of people towards wildlife since humans can affect wildlife and vice versa often resulting in conflict. These attitudes of people towards carnivores will be explored in chapter 6.

CHAPETER 6: REPORTED ATTITUDES OF PASTORALISTS TOWARDS WILDLIFE IN THE TARANGIRE ECEOSYSTEM

6.1 Summary

The aim of this chapter is to examine the attitudes of Maasai towards wildlife in the Tarangire ecosystem. I used semi-structured interviews to investigate factors which may shape attitudes of Maasai towards wildlife, namely: knowledge of wildlife species, diversity of income sources, direct benefits from wildlife, and the level of wildlife damage experienced. I also investigated the relationship between both the density of wild animals perceived to be a problem, and the distance to Tarangire National Park, on the intensity of reported conflict. Maasai in the Tarangire ecosystem reported higher levels of conflict with most species, particularly large carnivores, than reported elsewhere in Tanzania, despite relatively low levels of livestock predation. There were no reported losses of livestock from other causes, such as disease and theft, which may partially be due the restricted sample size of 83 agropastolists, and partially because the survey was only carried out during the dry season, which may affect disease prevalence. Few people received income from wildlife-related activities, which may help to explain the relatively high intensity of reported conflict in this area. Statistical analysis revealed that level of wildlife knowledge; number of small stock lost to predators, number of income sources and density of wild animals near human settlements perceived to be problematic were the key determinants of conflict. Given the intensity of conflict reported here and the biological and economic importance of the Tarangire ecosystem there is a clear need for conflict mitigation in this area. The development and implementation of WMAs could be important for lessening conflict, but the process must be driven and facilitated by local communities and other stakeholders, and lessons must be learned from the problems in successfully establishing of WMAs elsewhere in Tanzania.

6.2 Introduction

Understanding the attitudes of local communities towards wildlife is central to developing effective conservation strategies. Many people argue that the establishment of protected areas has negative impacts on local communities, especially in developing countries where livelihoods of local communities often depend on natural resource extraction (Ghimire and Pimbert 1997, Madden 2004). Negative impacts on local people's livelihoods include exclusion from traditional lands, limitations on resource access, and direct impacts of wildlife such as livestock predation and crop damage (Naughton-Treves 1998, Adams et al. 2004, Bauer and de Iongh 2005). However, there are also examples where the establishment of protected areas has been beneficial to local people. A recent study found that population growth rates near park boundaries in some parks in Africa and South and Central America was higher than national rural average growth rates (Wittemyer et al. 2008). This higher growth rate is thought to be due to perceived economic incentives as a results of conservation investments in and around parks, thus attracting more people to the area, although this increased population growth rate may in itself pose a threat to biodiversity conservation – the very reason for which the parks are intended (Wittemyer et al. 2008). However economic opportunities are not the only reasons that attract people towards parks. Other factors such as the presence of good infrastructure, employment opportunities, and development due to donor projects also play a role (Igoe et al. 2008). Nonetheless, given the limitations of protected areas discussed in chapter 1, such as rapid land use change outside protected areas and some reserves being too small to support wildlife populations throughout the year, then ensuring local people's support is extremely important for effective biodiversity conservation. However, many factors may affect this support including for example, the level of economic benefits received by local people (Newmark et al. 1993, Fiallo and Jacobson 1995). It is argued that local communities who receive benefits from wildlife-related activities are more positive towards wildlife conservation than those who don’t and many studies e.g. Gillingham and Lee (1999), Infield and Namara (2001) and Sekhar (2003) have demonstrated that such economic benefits are important for local people’s support to conservation. However, others have found that benefits to local communities do not necessarily lead to improved attitudes towards conservation: for instance, when there are inequalities in the distribution of economic benefits it may affect local people’s attitudes to conservation (Walpole and Goodwin 2001, Sachedina 2008, Sachedina and Trench 2009).

Another factor that appears to have a positive influence on people’s attitude to wildlife is relative wealth. Studies have shown that people with more diverse income sources tend to report less conflict with wildlife than people reliant upon few sources of income (Dickman 2008). This is not surprising because people with more income sources will be less vulnerable to wildlife damages or unexpected events (Naughton-Treves 1997, Cutter et al. 2000). Level of education is also another factor which has shown to have a positive influence on people’s attitudes towards wildlife. For example, in South Africa (Infield 1988) found that positive attitudes towards conservation increased with level of education. However, in some cases level of education can also influence negative attitudes - for instance, in southern Tanzania Dickman (2008) found that the intensity of reported conflict between people and wildlife increased with peoples’ level of wildlife knowledge.

Overall, many different factors influence negative attitudes towards wildlife and intensify human wildlife conflicts (Gadd 2005, Woodroffe et al. 2005, Parker and Osborn 2006, Browne-Nuñez and Jonker 2008). These conflicts have been defined as when the needs and behaviour of wildlife impact negatively on the needs of humans, or when the needs of people negatively impact on the needs of wildlife (Madden 2004). Such conflicts are particularly intense where livestock and agriculture are a major source of income, due to wildlife damage through livestock predation and crop damage (Naughton-Treves 1998, Naughton-Treves et al. 2003a, Madden 2004, Patterson et al. 2004, Thirgood et al. 2005, Zimmerman et al. 2005). Conflicts between humans and wildlife are also intense where it involves loss of human life (Packer et al. 2005).

A range of species cause conflicts with humans, from rodents to mega herbivores such as elephants (Hoare 1999). However, large carnivores are of particular interest, partially because of their intrinsic carnivorous behaviour, which put them into direct competition with humans for livestock and game species, but also because they are feared due to their ability to kill humans (Patterson et al. 2003, Packer et al. 2005, Baldus 2006, Røskaft et al. 2007). Several studies show that large carnivores are not responsible for as much damage as local people commonly perceive (Rasmussen 1999, Maddox 2003, Dickman 2005, Dickman 2008), yet the intensity of conflict with large carnivores is often high. This perception generates negative attitudes towards large carnivores, and means that many species are intensely persecuted by humans (Woodroffe and Ginsberg 1998, Woodroffe 2000a, Hussain 2003).

Sometimes negative attitudes towards wildlife may be driven by less direct causes, such as wildlife policy and legislation. For example, in the Loliondo area outside the Serengeti National Park, Maasai pastoralists rejected a government proposal for the implementation of a community run Wildlife Management Area, simply because local people saw the government's proposal as another way of state driven conservation practices and a threat to their traditional land (Nelson et al. 2009). In another study, Gillingham and Lee (1999) found that despite local communities agreeing to support wildlife conservation in the Selous Game Reserve, they had also negative views towards some of the activities of the wildlife institutions.

In the Tarangire ecosystem, Maasai people have coexisted with wildlife for many years (Igoe and Brockington 1999). However, the establishment of the Tarangire National Park in 1970, the introduction of large commercial farms by the government, the establishment of private farms and the development of mining have all led to reduced grazing areas and loss of customary land rights by Maasai (Mwalyosi 1991, Lama 1998). Furthermore, the population of the Maasai and non pastoral immigrants in the ecosystem has increased especially over the last three decades (TCP 1997, Igoe and Brockington 1999). This loss of customary land rights and the increase in human population has had significant impacts upon Maasai livelihoods (Mwalyosi 1992). Therefore, many Maasai in the ecosystem have now opted for cultivation in addition to keeping livestock in order to diversify their income sources (TCP 1997, Igoe and Brockington 1999). This expansion of cultivation in the ecosystem is viewed by many people as a move to protect the Maasai's traditional land rights (Sachedina and Trench 2009).

The aim of this chapter is to assess the magnitude of human-wildlife conflict in the Tarangire ecosystem, by examining socio-economic factors that may influence attitudes of Maasai towards wildlife, namely level of wildlife knowledge, diversity of income sources, wildlife-related benefits, and costs from wildlife due livestock depredation and crop damage. The main focus of this study is on carnivores although non-carnivores are also included. In order to achieve this aim, seven primary questions are addressed.

1. Does knowledge of species influence the intensity of reported conflict between people and wildlife?
2. Are some species considered more problematic than others?
3. Does relative wealth influence the intensity of reported conflict between people and wildlife?
4. Is intensity of reported conflict influenced by magnitude of stock losses?
5. Does monetary benefits from wildlife related activities decrease intensity of reported conflict between people and wildlife?

6. Is the intensity of reported conflict influenced by the density of problematic animals near human settlements?
7. Is the intensity of reported conflict influenced by distance to Tarangire National Park?

6.3 Materials and Methods

6.3.1 Data Collection

Semi-structured questionnaires (Appendix I) were used to collect data for this chapter, following the format used by Maddox (2003) and Dickman (2008) to survey pastoralists attitudes towards wildlife in northern and southern Tanzania respectively. A review of the methods is given in chapter 2. The survey assessed attitudes of Maasai towards wildlife in general. However, carnivores were the main focus for the interview (see also previous chapters), although this was not made explicitly to the respondents to avoid biased results. Data was collected to assess: (1) the knowledge and attitudes of respondents towards wildlife species e.g. their ability to identify species, and how problematic they were considered to be; and (2) whether certain socio-economic characteristics, such as levels of stock ownership, levels of stock losses to predators and other causes, the number of income sources and whether the respondents received any money from wildlife–related activities, affected views towards wildlife.

The survey targeted agropastoralists in three villages in the study area where camera trap surveys for carnivores and other wildlife species had been carried out in cultivated areas. Therefore respondents for this survey were only those whose farms had been surveyed for wildlife species and only targeted one ethnic group, the Maasai. Interviews were conducted by myself in Kiswahili and used a translator who spoke Maa when the need arose. The translator was a Maasai and a field assistant in the project. Interviews took approximately one hour to complete. All households were georeferenced using Global Positioning System for estimating distance to camera traps and to the Tarangire National Park for subsequent use in the analysis. Respondents were shown wildlife pictures (Appendix II) which included 30 African species and a tiger. The latter was used to check how reliable their responses were, given that the species does not occur in Africa. Respondents were asked to identify the animal and if they misidentified they were told the correct name. The level of correct identification was used as an index of wildlife knowledge. Respondents were also asked whether they would like to see the animals they identified in their farm or in the grazing areas near their boma and the reason for their response. They were also asked to rank the animals based on problem scores i.e. no problem = 0, small problem = 1 or a big problem = 2. For each identified species, respondents were also asked whether the species occurs in their area i.e. within a day's walk, and the reason why they felt it was a problem. The problem scores were used in a comparative analysis with those found by Dickman (2008) with Maasai in southern Tanzania. Other data collected during the survey were on income sources. Respondents were asked if any member of their family received cash from livestock, crops, off-farm work or wildlife related activities e.g. photo tourism and hunting. Respondents were also asked to rank their sources of income in the order of importance to their livelihoods. Other questions that were asked included: the number of cattle, small stock (sheep and goat), donkeys, and chickens that were sold, slaughtered, given away, killed by predators, or died from other causes, in the one-month period preceding the interview.

6.3.2 Data Analysis

Data was carried out using Statistical Package for Social Scientists version 14.0 (SPSS Inc). I used one sample Kolmogorov-Smirnov tests to test for normality in the data, and used non-parametric methods where the data did not conform to a normal distribution. Parametric statistics were used when the data which conformed to a normal distribution. Tests used included the Kruskal-Wallis H test, Pearson's correlation, Wilcoxon test, univariate analysis of variance, linear regression and hierarchical cluster analysis. All statistical tests were two-tailed with the level of significance defined as $P < 0.05$.

6.4 Results

6.4.1 Respondent's Characteristics and Demography

A total of 83 interviews were conducted in three villages outside the Tarangire National Park in Simanjiro and Monduli districts (Table 13). The number of surveys conducted differed between villages because it was dependent on the number of farms where camera traps had been placed during the previous wildlife surveys. The majority of the interviews (60.2%) were conducted in Loiborsoit village, 32.5% in Lokisale and 7.2% in Emboreet. Respondent age ranged from 27 to 74 years old, with a mean age of 46.6 (SD $\pm$ 9.8).

Table 13: Sampling effort for three villages visited during interviews in the Simanjiro and Monduli districts

Villages	Interviews	District
Loiborsoit	50	Simanjiro
Emboreet	6	Simanjiro
Lokisale	27	Monduli
Total	**83**	

6.4.2 Wildlife Knowledge

Respondents correctly identified an average of 16.8 (range 11 - 23) of the 30 African species shown. None of the respondents identified the tiger, which they claimed did not occur in their farms and in the grazing areas, and could not describe where the tiger might occur. The level of species identification varied between species. Generally, species such as giraffe, elephant, buffalo, impala, lion, leopard, spotted hyaena were identified correctly by 100% of all respondents, but other species, particularly nocturnal or smaller ones, had low level of identification. For example, the common genet, zorilla and aardwolf were identified by less than 20% of the respondents (Figure 14). There was no significant relationship between the age of respondents and the level of species identification (r_s = - 0.061, n = 83, P = 0.585).

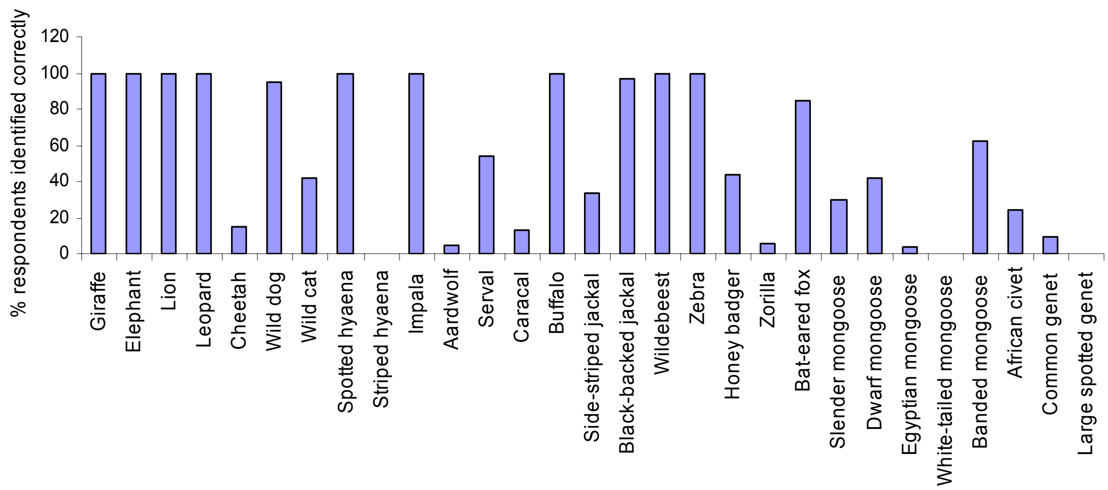

Figure 14: Percentage of respondents that identified photographs of African wildlife species.

6.4.3 Relative Wealth and Wildlife-Related Benefits

6.4.3.1 Household Income Sources and Diversification

All respondents (n = 83) had at least one strategy for generating income. The number of income sources ranged from 2 - 4, with a mean of 2.5 (SD $\pm$ 0.7). Reported income sources are shown in Table 14. None of the people interviewed said they received any income from remittance (money sent to respondents from relative etc. living outside the village). Thirteen people (15.7%) reported that they received income from trophy hunting by working as guides and watchmen at hunting camps. Two of these people lived in Emboreet while 7 lived in Loiborsoit and 4 were from Lokisale. Four people (two from Loiborsoit and two from Lokisale) also reported receiving income from photo tourism by working at tourist camps. Overall, 20.5% of the people interviewed received some income from wildlife related activities.

Table 14: Reported income sources of interviewees

Sources of income	n	Percent
Livestock	80	96.6
Crops	83	100
Trophy hunting	13	15.7
Photo tourism	4	4.8
Remittance	0	0
Other sources	36	43.4

Although all people (100% n = 83) were engaged in agriculture as one of the sources of income (Table 14), analysis showed that livestock was the most important source of income for the majority of people (71.1%, n = 59) (Table 15, see also Figure 15). Agriculture was the main source of income for only 19.3% of the respondents (n = 16) and only 9.6% of the interviewees reported other sources such as mining and formal employment as their main source of income which included mining, formal employment e.g. two of the respondents were employed in a primary school, one as a teacher and the other as a cook. Other sources of income reported included photographic and wildlife hunting tourism and small scale business, such as selling beads and tractor rental for farm cultivation.

Table 15: Reported main income sources of interviewees

Source of income	n	Percent
Livestock	59	71.1
Crops	16	19.3
Other sources	8	9.6

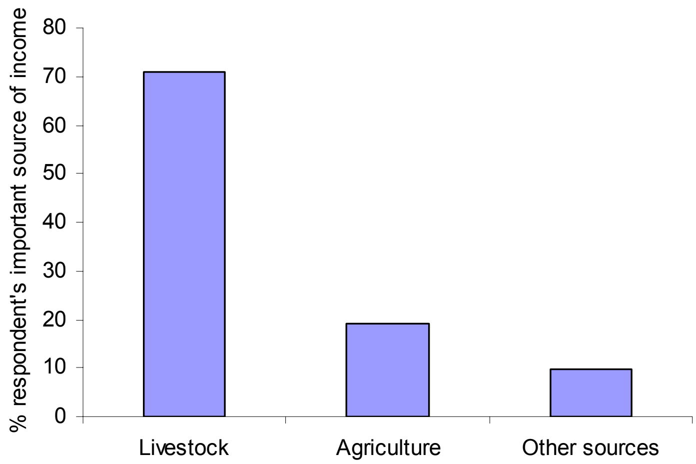

Figure 15: Reported main sources of income by Maasai in Loiborsoit, Emboreet and Lokisale villages

6.4.3.2 Stock Ownership and Composition

Respondents owned between 10 and 300 livestock at the boma visited, with a mean of 84.9 (SD $\pm$ 78.7) livestock at each household. Small stock (sheep and goats) were kept by all respondents but were not the most abundant stock owned. Small stock accounted for 26.5% of all stock holdings. Cattle were the most abundant stock owned, accounting for 65.2% of all stock holdings, while donkeys accounted for 4.3% of stock holdings and chickens were accounted for 4% (Table 16, see also Figure 16).

Table 16: Reported stock composition by Maasai

Parameter	**n**	**Percent**	**Percent total stock**
Cattle	82	98.8	65.2
Small stock	83	100	26.5
Donkeys	82	98.8	4.3
Chickens	52	62.7	4

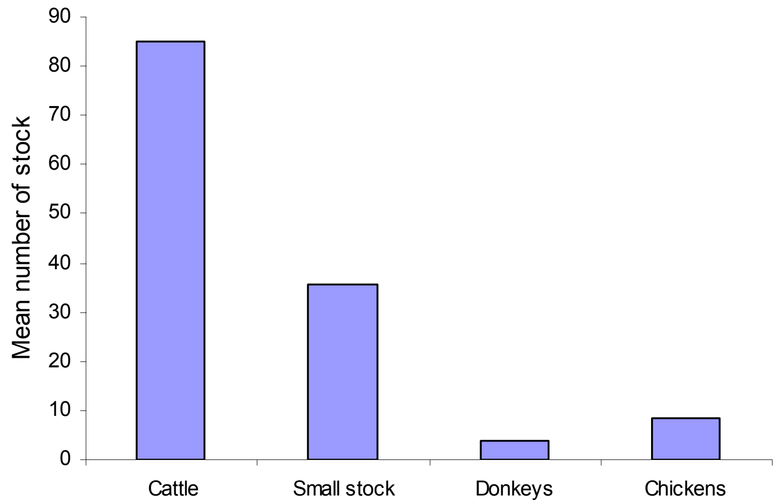

Figure 16: Reported level of stock ownership by Maasai

6.4.4 Livestock Use and Losses

6.4.4.1 Reported Levels of Stock Use and Losses

The majority of the respondents (67.3%, n = 83) used at least one animal for selling, slaughtering or giving away as gift the month before this survey was carried out, while only a few people (7.2%, n = 83) reported to have lost at least one animal to carnivores with an overall mean of 0.11 and average loss of 0.08 animals for each respondent. All reported losses were from small stock only (Table 17).

Table 17: Level of livestock use and losses reported by Maasai

Parameter		Cattle		Small stock		Donkeys		Chickens		Overall mean	
		Mean no.	Mean % of herd size	Mean no.	Mean % of herd size	Mean no.	Mean % of herd size	Mean no.	Mean % of herd size	Mean no.	Mean % of herd size
	Sold	0.77	0.91	0.58	1.62	0.00	0.00	0.22	2.66	1.96	1.30
	Slaughtered	0.14	0.16	0.00	0.00	0.00	0.00	0.03	0.36	0.04	0.13
Stock use	Given away	0.43	0.51	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.13
All stock uses		**1.34**	**1.58**	**0.58**	**1.62**	**0.00**	**0.00**	**0.25**	**3.02**	**0.54**	**1.56**
	Stolen	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Killed by predators	0.00	0.00	0.11	0.31	0.00	0.00	0.00	0.00	0.03	0.08
Stock loss	Died	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
All stock losses		**0.00**	**0.00**	**0.11**	**0.31**	**0.00**	**0.00**	**0.00**	**0.00**	**0.11**	**0.08**

6.4.4.2 Reported Causes of Stock Loss

Predators were the reported cause of stock loss during the month prior to this survey. Leopard and spotted hyaena were reported as the main cause for small stock losses with 66. 7% of the losses caused by leopard and 33.3% caused by spotted hyaena.

6.4.5 General Attitudes towards Wildlife

When asked about whether they will be happy to have wildlife in their farms and in the grazing areas, the majority (84.3% n = 78) of the people said they were not happy to have wildlife species in their farms and in the grazing areas and very few (15.7%, n = 5) said they will only be happy if they receive tangible benefits. The reason given for those who disliked wildlife was predation of livestock and crop damage caused by wildlife.

6.4.5.1 Conflict Scores with Wildlife Species

The mean conflict scores for all the 30 African species was 1.05 ($\pm$0.19), with significant variation between different species (KW $\chi^2 = 622$, $df = 2$, $P < 0.001$; Figure 17). Large carnivores were ranked as significantly more problematic than non-carnivores ($z = -7.9$, $P < 0.001$).

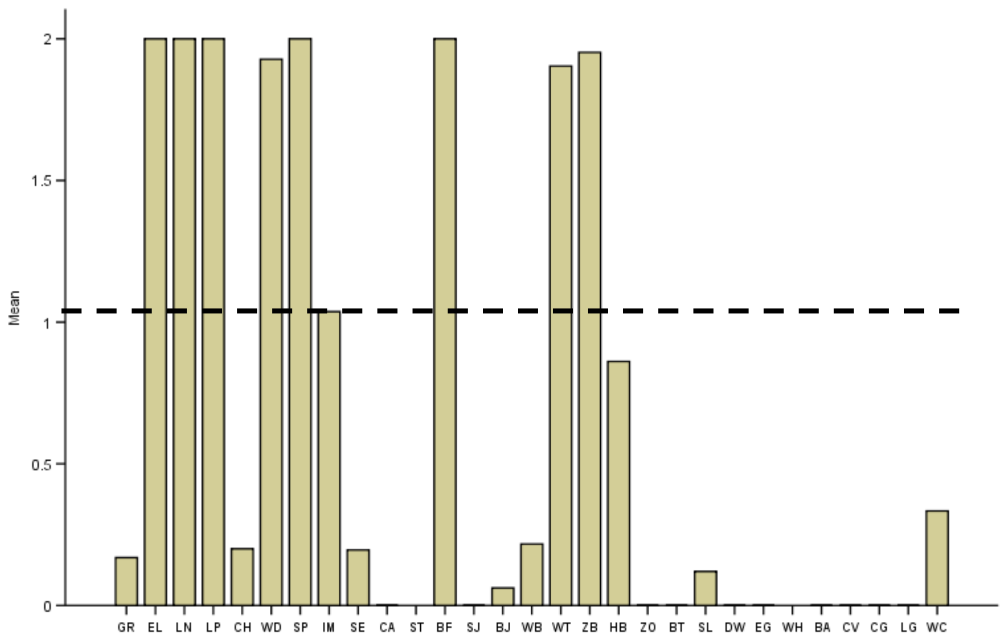

Figure 17: Mean conflict scores assigned by respondents to the African wildlife species that were shown during the survey. The dotted line indicate mean across all species

6.4.5.2 Conflict Clusters

A hierarchical cluster analysis on the reported conflict scores was carried out in order find out which had similar problems. Results revealed three distinct clusters in terms of their degree of perceived conflict with respondents (Figure 18). Cluster 1 consisted of species with low conflict scores, all of which were medium and small carnivores, while Cluster 2 contained of species which caused small-scale problems both carnivores and non-carnivores. Cluster 3 comprised of species which were frequently reported as a threat to livestock and/or people and crops. Four large carnivores, namely lion, leopard, African wild dog, and spotted hyaena, as well as buffalo, warthog, zebra and elephant were grouped into this high-threat cluster. Although elephant, lion, leopard, spotted hyaena, buffalo, zebra, warthog and wild dog were rated similar on average conflict score above, when people were asked which were the most problematic species, results showed that lion, leopard, spotted hyaena and zebra were most problematic followed by wild dog and warthog (Figure 19).

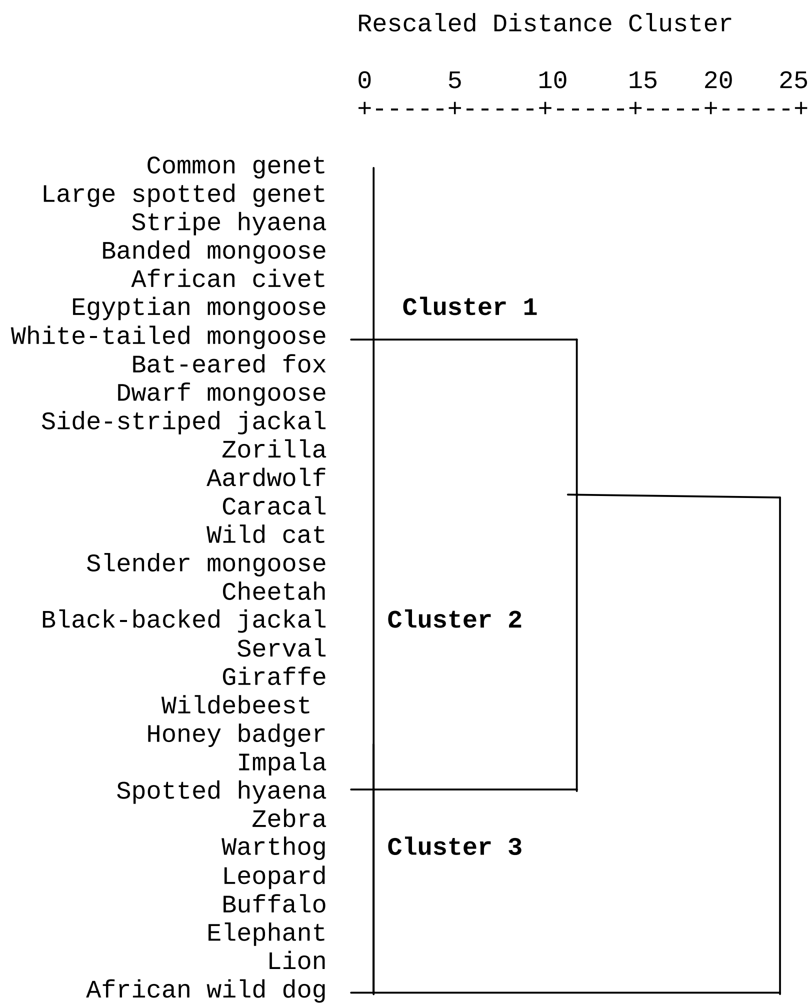

Figure 18: Dendogram of hierarchical cluster analysis showing three clusters based on average distance of conflict scores between respondents.

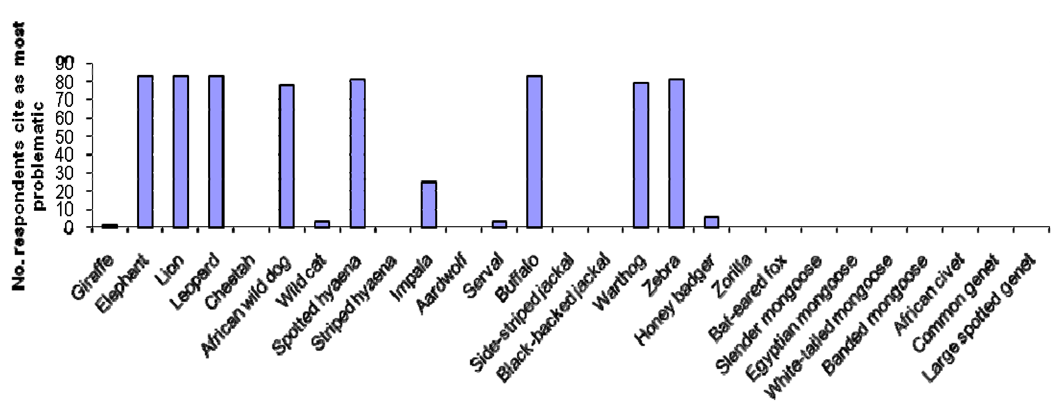

Figure 19: Species named as most problematic by respondents

6.4.5.3 Comparison of Conflict Scores with Elsewhere in Tanzania

Dickman (2008) documented high levels of conflict between pastoralists and wildlife in southern Tanzania, focusing on five ethnic groups, namely the Maasai, Barbaig, Hehe, Bena and Sukuma. A comparison of the scores found in this study with her results with the Maasai in southern Tanzania shows that, across all respondents, conflict scores in the Tarangire ecosystem were higher than those found in Ruaha, apart from the cheetah, striped hyaena and serval which had higher conflict scores in Ruaha (Figure 20).

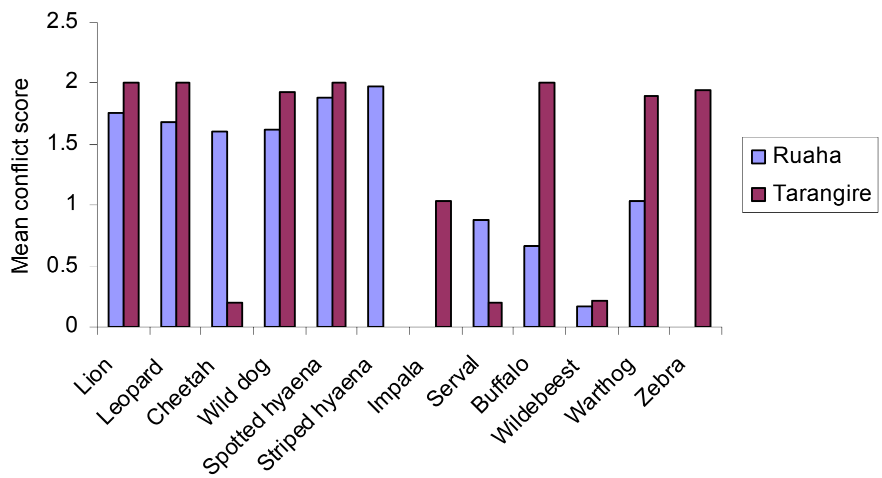

Figure 20: Comparison of reported conflict scores found by Dickman (2008) for Maasai in southern Tanzania with those found in this study for all respondents and for Maasai alone. Higher conflict scores indicate high level of perceived problem.

6.4.5.4 Key Determinants of Conflict with Wildlife

Results showed that levels of reported conflict increased with the level of wildlife knowledge exhibited by respondents (r = 0.57, df = 81, P < 0.001) and the number of small stock loss (r = 0.28, df = 81, P = 0.009) but not the total number of stock owned (r = 0.05, df = 81, P = 0.671). Intensity of conflict varied between people with different primary income sources (F = 2.66, df = 3, P = 0.05), conflict increased with the number of income sources (r = 0.28, df = 81, P =

0.011) Furthermore, intensity of conflict increased with the number of species perceived to be a problem that were photographed at camera stations near human settlements (r = 0.259, df = 81, P = 0.01) but not with distance to the park (r = 0.09, df = 81, P = 0.434).

6.0 Discussion

Many wildlife species were considered problematic by Maasai in the Tarangire ecosystem, with reported conflict scores higher than those documented elsewhere in Tanzania. Intensity of conflicts among people increased with diversity of income sources contrary to previous findings. A number of studies have shown that relatively wealthy people are often more tolerant of wildlife damage than poor people (Oli et al. 1994, de Boer and Baquete 1998), because a diversity of income sources reduce vulnerability to wildlife-related damages (Naughton-Treves 1997, Cutter et al. 2000). The reason why intensity of conflict increased with diversity in income sources here is unclear. However, it shows that other factors are likely to be important in driving the reported conflict e.g. the low level of benefits that people are receiving from wildlife in this area. Interestingly, although the reported number of livestock killed by predators was low, large carnivores were considered to cause significant problems. Similar results have also been obtained from several previous studies e.g. Rasmussen (1999), Maddox (2003) and Dickman (2005, 2008) found that perceived conflict between livestock keepers and large carnivores was often higher than actual losses caused by predators probably, because of the fear among people that large carnivores pose a threat not only to livestock but also to human life.

It was surprising that this study found no livestock losses due to other causes such as disease and theft. Lack of losses from other causes such as diseases may be due to the fact that the sample size was restricted to agropastoralists where wildlife surveys had previously been carried, out but also because the interviews were carried out during the dry season only.

Livestock losses from causes other than predation are common among pastoralists - for instance, diseases such as East Coast fever has been reported to cause high calf mortality (60% annually) among Maasai pastoralists and agropastoralists depending on rainfall (Homewood et al. 2006). Similarly in Emboreet village in the Simanjiro District, it has been reported livestock diseases are generally higher during the wet season because of higher tick loads, and many Maasai believe that rain dilutes the effects of accaricides (Sachedina and Trench 2009). However it has also been reported that diseases such as Malignant catarrhal fever and nose bot have declined in the area because of declining numbers of wildlife calving in the Simanjiro Plains (Sachedina and Trench 2009). Studies have also shown that theft causes substantial losses among livestock keepers (Dickman 2005, Dickman 2008). Lack of losses from theft found here could be due to the small sample size of respondents but also it might be that theft is less common in the region, probably because there are no other major pastoral ethnic groups in the area. Livestock theft is most common between different pastoral and agropastoral ethnic groups is Tanzania, for instance between Kuria and Sukuma (Rodgers 1992), or between Maasai and other ethnic groups (Brockington 2004).

Wildlife benefits to local communities can reduce conflict between people and wildlife and encourage support for conservation (Oli et al. 1994, de Boer and Baquete 1998). This concept assumes that benefits will provide economic incentives for communities to tolerate coexistence with wildlife by addressing poverty and conservation (Wells and Brandon 1992). However, in many developing countries wildlife benefits rarely offset the economic losses incurred by the rural poor (Ghimire and Pimbert 1997). This is because benefits from wildlife and biodiversity conservation often go to governments and external entrepreneurs, while many of the costs are actually borne by individuals at household level (Dixon and Sherman 1990, Balmford and Whitten 2003). Under such circumstances, it is unlikely that local communities will improve tolerance of wildlife, as a result of which wildlife population continue to decline in many areas through persecution and habitat destruction (Walpole and Thouless 2005). In the Tarangire ecosystem, community-based conservation activities have been established by the park authorities in villages outside the park in order to ensure that local people benefit from wildlife conservation (Sachedina 2008). This study found that few people benefited from the wildlife activities, which is probably why the intensity of conflict reported was high. However, it is possible that the low number of people who reported benefited from wildlife activities here reflected the relatively small samples size, as we only targeted agropastoralists whose farms were surveyed for wildlife species. Nonetheless in another study Sachedina (2008) also found that only few elite people benefited from wildlife activities, suggesting that wildlife-related benefit-sharing is indeed generally low in the area. Furthermore, Sachedina (2008) also found that more households that received wildlife income increased their area under cultivation compared to those who didn't receive any wildlife income. Many people suggest that although cultivation is seen as a livelihood diversification strategy among pastoralists in the Simanjiro, it is also a means of acquiring land and protecting it from possible expansion of conservation areas (Sachedina 2008, Sachedina and Trench 2009).

The finding that conflict levels increased in tandem with the density of problem species near households and respondents' knowledge of wildlife species was not surprising. This because people knew which problem species were found near their homes and the threat they could pose to their livestock, crops or human life and therefore they had negative attitudes towards those species. Previous studies have also reported a increase in intensity of conflict with the level of wildlife knowledge (Dickman 2008). The presence of high densities of problematic animals near human settlements means that there is high probability of them encountering livestock, people or crops and therefore generating conflict with people. However, the intensity of conflict was not influenced by the distance to the Tarangire National Park. It was expected that intensity of conflict between people and wildlife would be higher near the park because wildlife is likely to be more abundant in the Tarangire National Park (see chapter 4), especially during dry season when most migratory species return to the park (Borner 1985, Kahurananga and Silkiluwasha 1997, TCP 1997, Sachedina 2008) . The reason why intensity of conflict was not influenced by distance to the park is not well understood, and future investigation will be important to understand this relationship.

Overall results of this study showed that wildlife knowledge, livestock predation, low levels of wildlife-related benefit-sharing and density of problem animals near human settlements may be the key factors underlying the reported high conflict with wildlife, particularly large carnivores. It is common, especially in developing countries, that the establishment of protected areas often involve the exclusion of people (McNeely and Miller 1984, Western and Wright 1994). This exclusion of people affects local people's livelihoods and generates negative attitudes towards protected areas because of the opportunity costs i.e. the benefits that local people could have received if the area was not designated as a protected area (Salafsky and Wollenberg 2000). Pastoralists in northern Tanzania are among the people whose livelihoods have been transformed by the creation of protected areas (Homewood and Rodgers 1991, Neuman 1998, Igoe 2004). In the Tarangire ecosystem, pastoralists were evicted when the Tarangire National Park was established (Igoe and Brockington 1999) and therefore grazing areas and access to water were reduced. Grazing areas were reduced further due to the introduction large scale commercial farming by government and private investors and mining (Mwalyosi 1991, Lama 1998) but also through small scale allocation by village governments to in-migrants in the area (Igoe and Brockington 1999). Furthermore, increase in the population of both Maasai and non-pastoralists immigrants has increased demand for land (TCP 1997). According to (Muir 1994) the combined effect of loss of pastoral rangelands and the growing human populations as well as poor livestock health services have resulted in the decline of per capita herd size. Consequently cultivation is now seen by Maasai as an alternative sources of income (Igoe and

Brockington 1999), but also most important as means to protecting their land from further land loss to wildlife conservation (Sachedina 2008, Sachedina and Trench 2009). The expansion of agriculture has elicited concern among conservationists because cultivation is thought to result in decline of wildlife populations (TCP 1997, Sachedina 2008, Sachedina and Trench 2009). With this past experience, the attitudes expressed by the Maasai towards wildlife in this study may represent their resentment to wildlife conservation in the Tarangire ecosystem.

Given the biological and economic importance of the Tarangire ecosystem, there is a need to have concerted effort for addressing both the needs for wildlife conservation and the protection of local community livelihoods in order to promote coexistence. One possible solution would be to promote the establishment of WMAs in the region, which involves participatory land use planning. However, this can only be achieved if the process is driven by local communities rather than outsiders, and governments and other stakeholders must facilitate the process and carefully consider the problems that have been encountered in WMA establishment elsewhere in the country.

CHAPTER 7: GENERAL DISCUSSION

The aim of this thesis was to examine the impact of human activities, specifically to test the use of camera traps for monitoring mammal biodiversity; to examine the impact of land use change on carnivores and non-carnivore prey; and to identify drivers of conflict between Maasai and carnivores. Here I present my main findings, including their broad implications for conservation of biodiversity, and indicate areas for future research.

7.1 Monitoring Mammalian Biodiversity

Monitoring of biodiversity is essential for effective conservation planning because it provides information on species distribution which conservation managers and policy makers need for setting conservation priorities (Maillard et al. 2001, Linkie et al. 2006, Georgiadis et al. 2007). Monitoring of carnivores is particularly important when compared to other taxa because the population of most species in this taxa are declining very fast in many parts of the world due to habitat loss, hunting, diseases, depletion of prey species and trade in body parts (Novaro et al. 2000, Ray et al. 2005, Karanth and Chellam 2009). However this fast decline in carnivore populations is also attributed to inherent biological factors which make carnivores to be more vulnerable both to natural and anthropogenic factors e.g. many species of large carnivores have large home ranges and thus require large habitats to survive and many occur at relatively low densities (Woodroffe 2000a, Cardillo et al. 2004). The conservation of carnivores can be very important e.g. being at the top of many food chains carnivores are important in maintaining ecosystem health (Terborgh et al. 1999, Terborgh et al. 2002, Ray 2005). They are also often used as umbrella species - species that need large tracts of habitat, therefore by conserving such species many other species are automatically saved (Simberloff 1998) (see also chapter 1 for details) and where carnivores are tame and conspicuous they can bring economic benefits through photographic tourism especially to developing countries (Treves and Karanth 2003).

A range of methods exists for monitoring mammal biodiversity e.g. direct methods such as line transects and aerial count and the use of indices e.g. spoor and dung counts. Interviews are also commonly used especially for broad scale assessment and more recently camera traps have been used to estimate abundance of individually recognisable mammals and relative abundances. Despite the availability of a range of techniques for monitoring mammals, most of these are unlikely to be effective for monitoring carnivores because carnivores are shy, some are nocturnal, and many especially large carnivores tend to occur at low densities (Smallwood and Fitzhugh 1993, Ogutu and Dublin 1998). However the use of camera traps has shown to be effective in monitoring a range of mammal under a wide range of environments e.g. for inventorying species (Silveira et al. 2003, Tobler et al. 2008, Pettorelli et al. submitted), estimating abundance of individually recognisable species (Karanth and Nichols 1998, Silver et al. 2004) and for determining mammal relative abundances e.g. photographic rates (Carbone et al. 2001, Yasuda 2004).

In this study the use of camera traps has shown to be effective in monitoring mammal biodiversity, particularly for carnivores when compared to most conventionally available wildlife monitoring methods. There are a few other methods that can monitor carnivore effectively, and for large mammals e.g. elephants, kudu, zebra, eland, and wildebeest they can be monitored effectively using aerial counts. Camera traps were successfully used to assess species richness, relative abundance, and absolute abundance of individually recognisable species for ground dwelling medium to large mammals. The broad applications of camera traps for monitoring mammals and its limitations are discussed below.

7.1.1 Species Richness

My results suggest that camera traps were effective in assessing mammal species richness in the Tarangire ecosystem. A total of 18 carnivores and 23 non-carnivores were recorded during this study, a diversity level which would not have been possible with commonly available methods for mammal inventories. In illustration of this point, the surveys recorded bushy-tailed mongoose for the first time in the Tarangire ecosystem, despite a substantial amount of survey work using other methods (e.g. aerial surveys and line transects) prior to this study. Previous published records of this species in Tanzania have only been from southern dry and moist forests and along the coastal thickets (Kingdon 1997), however recent camera trap surveys including this one have recorded the species beyond this range (Pettorelli et al. submitted). Other studies have also shown that camera traps were more effective in inventorying ground dwelling mammals when compared to line transects (Silveira et al. 2003, Tobler et al. 2008). Whilst line transects may work well for inventorying larger species of mammals in open habitats where animals are more conspicuous, the method is less effective for smaller, shy and elusive or low density species as is the case with some carnivores (Stander 1998, Bashir et al. 2004).

Interviews is another commonly used technique for documenting the status and distribution of many species on broad scales such as national or continental e.g. the IUCN Carnivore Action Plans (Ginsberg and Macdonald 1990, Nowell and Jackson 1996), but the quality of information gathered through interviews is often very limited (Mills 1997). The method has mainly been used for large and medium-sized species probably because large and medium-sized species are more likely to be unambiguously identified (Ginsberg and Macdonald 1990, Nowell and Jackson 1996). Interviews can yield unreliable results for species which are likely to be confused with other species by local people. For example, in a survey conducted using interviews in south-eastern Pantanal, Brazil there was uncertainty on occurrence of three species, three-banded armadillo (*Tolypeutes matacus*), bush dog (*Speothos venaticus*) and dwarf brocket (*Mazama nana*) that were reported by local people. It was found that the three species were likely to confused by the local people (Trolle 2003). Interviews are therefore unlikely to be effective for inventorying rare, elusive and nocturnal and smaller species, as is the case for many carnivores (Tobler et al. 2008).

Results demonstrated that time required for assessing species richness varied between taxa. Comparison of species richness between carnivores and non-carnivores in this study revealed extensive survey time (75 days) was required to record the majority of species, particularly carnivores. This is probably because carnivores are generally cryptic and often occur at low densities. Rarefaction curves - which show the expected species richness based on the observed species richness curves - showed that, given the same amount of trapping effort, most carnivore rarefaction curves did not reach asymptotes or even start levelling off, regardless of land use type whereas for non-carnivores most of them had started levelling off. This suggests more time was needed for carnivore species to be recorded than non-carnivores. This difference in captures is important for designing monitoring programs targeted at each taxon, because it shows that different taxa may require different sampling intensities.

Although the use of camera traps has shown to be effective for inventorying mammals, there were also some limitations associated with the technique. Camera traps were most appropriate for medium-sized and large mammals which can generate sufficient heat for the camera sensor to detect and take photographs. For this study non-typical DeerCam 300 cameras were used and were not suitable for taking photographs of small mammals such as rodents and shrews, although in recent years the introduction of new models of digital camera may make it possible to detect a wider range of mammal sizes in future. The use of camera traps outside protected areas can also be a problem due to theft. During my surveys a total of 29 cameras were stolen in the pastoral grazing areas out of which eight were stolen in cultivated areas. Although I was able to replace cameras and continue with the surveys, the data by way of exposed films was often lost alongside these cameras, reducing the survey effort in the grazing areas. The reason why few cameras were lost in the cultivated areas might be because of the fact that we had requested owners of the farms to take care of the cameras and were rewarded at the end of the survey if cameras were safe. Sometimes fire incidences can also be a problem – five cameras were lost to a fire that got out of control in the park during early burning activities which are part of park fire management practices. Furthermore, although all cameras were checked before installation, in some cases camera traps may not have functioned properly. In the Tarangire National Park some cameras occasionally took blank photographs even when there was no animal movement along the trail. The reasons for this are not fully understood, but factors such as the sun heating up vegetation in front of the camera combined with movement driven by wind might be expected to trigger the sensors on the cameras. Another limitation is that the initial costs may be relatively high e.g. for this study a total of 100 camera units were purchased, costing about \$ 8,000. Running costs such as batteries and films, can be expensive, and add to the costs of implementing a survey. However, whilst line transect methods are likely to be cheaper in more open habitats, other methods, such as aerial survey, may be more expensive.

7.1.2 Relative Abundance

The study demonstrated that camera trap photographs could be used to determine species relative abundances i.e. indirect measure of species abundance (index) that can be used to monitor species abundance. Quite often trends in numbers are what wildlife managers want to know rather than overall density. This is because trends in numbers can provide early warning about the status of species and therefore allow wildlife managers to develop management prescription (Maillard et al. 2001, Linkie et al. 2006, Georgiadis et al. 2007). In this case relative abundance estimates may be important for two reasons. First, they can be used where direct methods such as distance-based sampling (Buckland et al. 1993) or aerial counts (Norton-Griffiths 1978) are likely to fail e.g. for species that occur at low density as is the case with most large carnivores or for smaller and cryptic species direct methods may not be effective (Smallwood and Fitzhugh 1993, Ogutu and Dublin 1998). Direct methods can also be expensive and time consuming (Smallwood and Fitzhugh 1993, Ogutu and Dublin 1998). Second, estimating species relative abundance may be relatively cheaper and less time consuming and does not require identification of individuals (MacKenzie et al. 2002, O'Brien et al. 2003, Yasuda 2004), but also often wildlife managers require monitoring techniques that are efficient, cost-effective and provide timely results for developing conservation plans (Ogutu and Dublin 1998, Linkie et al. 2006).

Relative abundance estimates from camera traps can be more effective for monitoring species compared to other available methods such as spoor or track counts. For example, it would be very difficult or even impossible to determine relative abundance for the number of species that this study was able to survey using spoor count. Spoor count may be effective for large mammals but not smaller species like mongoose that were surveyed in this study. Tracks of large mammals such as elephants are more conspicuous and are relatively easy to identify but also can stay longer on the substrate compared to smaller species such as mongoose and therefore spoor or tracks counts have widely been used to census large carnivores e.g. leopards (Stander 1998), mountain lions (Smallwood and Fitzhugh 1993) and large carnivores (Gusset and Burgener 2005). The use of spoor counts requires experienced trackers who may not be easily available (Stander 1998), whereas it is relatively easy to identify species from camera trap photographs when compared to tracks.

Vocalisations are also commonly used to provide and index of abundance of animals. The method is particularly used to census large carnivores which vocalise or respond to calls (Long et al. 2008). This technique therefore may not be effective for monitoring carnivore community as was the case for this study. Vocalisation has been used frequently to survey spotted hyaenas, lions and wild dogs (Ogutu and Dublin 1998, Maddox 2003, Robbins and McCreery 2003). The bias towards large carnivores is probably because these species are the best competitors and are more likely to be able to scavenge from kills (Creel and Marusha 1996), which makes them more likely to be attracted to vocalisation. The method however, is limited in carnivore survey because not all sex and age classes are equally attracted to vocalisation (Windberg and Knowlton 1990). Furthermore it has been shown that if animals become habituated, their response to vocalisation tends to decline and therefore may under estimate abundance (Ogutu and Dublin 1998). Vertical air temperature and velocity of wind and availability of prey species also influences response to vocalisation e.g. it has been demonstrated that response of lion to vocalisation decreased with increasing abundance of wildebeest whereas spotted hyaena response did not vary with prey abundance (Ogutu and Dublin 1998).

Another method that is commonly used to estimate species relative abundance is the use of dung count. Dung counts provides index of species abundance. This technique is commonly used in the African savannahs for censusing elephants (Shorrocks 2007) e.g. it has been used successfully to census elephants in Nzinga Game Ranch in Bukina Faso where it was shown to be more effective when compared to aerial count (Jachmann 1991). Similarly dung count was also shown to be most effective for censusing mammals in the Hluhluwei-iMfolozi park in South Africa when compared to observational counts (Cromsigt et al. 2009). Despite this success story, the technique may not be effective large mammal community like the one found in the Tarangire ecosystem because it can be labour intensive but also sufficient time may be needed to provide training in identification techniques of dung from various species and may be quite challenging for smaller species such as mongoose.

It is clear from the discussion above that most of the commonly used methods that are used to determine index of mammal abundance cannot be effective for monitoring large mammal community as the one found in the Tarangire ecosystem. The use of cameras traps to provide relative abundance seems to be most effective and has the potential to be applied widely.

7.1.3 Species Absolute Abundance

Often, it is important to have estimates of species density for developing conservation and management plans. Particularly in hunting where sustainable off takes need to be calculated. In chapter 5, I showed that five individually recognisable species (leopard, serval, common genet, aardwolf and spotted hyaena) were photographed, and density estimates was determined for leopard, serval and aardwolf, but not for common genet and spotted hyaena due to a lack of sufficient recaptures. The reasons why some species could not be recaptured either between seasons or within season is variable. For some species availability of travel routes during the wet and dry season or availability of prey species may have been important. Camera trap spacing had a clear influence on recapture rates for smaller species such as serval, which are likely to have smaller home ranges.

Generally these results show that the use of capture-mark-recapture techniques within a camera trap survey framework is effective for estimating the abundance of some individually recognisable carnivores in protected areas. However the effectiveness of the technique outside protected needs to be tested. This is because effectiveness of capture-mark-recapture in camera trapping partly depends on the density of target species which influences likelihood of detection which is generally higher when the species density is higher (Karanth and Sunquist 2000), and therefore because density of animals may vary between land use types it is important to test effectiveness of the technique. It is important also to understand, that the use of camera traps and capture-mark-recapture techniques to estimate abundance of mammals in the wild is limited to species that can be identified individually. For the majority of species other species other methods may be used e.g. line transects have been used to census spotted hyaenas, jackals and lions in the open grassland habitats on the Serengeti plains (Hofer and East 1995) and Durant et al., (in prep). Furthermore, the use of line transects have also been effective in estimating the abundance of large mammals in the Tarangire – Manyara ecosystem (Msoffe et al. 2007). Generally, line transects are most suitable for species which are more conspicuous such as large mammals and in open habitats (Bashir et al. 2004, Sutherland 2006). Although the method is commonly used there is no general agreement on its cost implications. Some people suggest that line transect can be quite expensive (Msoffe et al. 2007) but relatively cheaper in more open habitats (Bashir et al. 2004). Overall the costs of the method depends on the target species and the area being surveyed and other logistics e.g. availability of trained personnel or potential people to be trained. Generally the rarer the species and the more difficult it is to detect it, the more kilometres need to be driven, which can increase the cost to the point where it is prohibitive or not feasible.

Aerial surveys are frequently used to estimate the abundance of large ungulates in savannah habitats. But the method is generally expensive, and is not suitable for species below the size of a Thomson's gazelle or in forested habitats (Msoffe et al. 2007). Furthermore the technique also generates large standard errors. Nonetheless, it has been successful in picking up large scale changes in large herbivore abundance in Tanzania (Stoner et al. 2007).

Overall it can be concluded that while the use of camera traps to estimate mammal abundance seems to be effective, however the technique is limited to species that can be individually recognised. The species abundances estimated here are the ones which are not easily estimated through other means and in fact there is no other estimate for densities of serval, leopard and aardwolf for the Tarangire ecosystem, and, indeed, few such estimates for any areas in Tanzania. Other methods such as line transect and aerial survey may be useful for some other species, depending on the objective of the study, although aerial surveys in particular, can be very expensive.

7.2 Relationship between Land Use and Mammal Biodiversity

Data showed that species richness and abundance varied between land use types. There was no significant difference in carnivore species richness between the park and pastoral grazing areas outside the park, but species richness was significantly lower in cultivated areas. For non-carnivores, species richness was higher in the pastoral grazing areas than in than in the park and in the cultivated areas. This variation was not independent of body size, and generally most large bodied mammals were absent from the cultivated areas. Despite the similarity in carnivore species richness between the park and pastoral grazing areas, the relative abundance of mammals was markedly higher in the park than in the pastoral grazing areas and significantly lower in the cultivated areas.

The pattern of species richness and abundance between the three land use types observed here has important implications for conservation of mammal biodiversity, as it suggests that mammal biodiversity decreases with increasing human impacts. The lower species richness and abundance found in the cultivated areas provides evidence that environmental change may have impacts on mammal biodiversity. Land use change influences biodiversity by altering habitats, ecological processes and biotic interactions and may reduce suitability of habitats for biodiversity conservation (Marzluff 2001). Studies suggest that land use change is one of the greatest threats to biodiversity throughout the world (Vitousek et al. 1997, Sanderson et al. 2002, Burgess et al. 2007, Jetz et al. 2007). However the higher non-carnivore species richness in the pastoral grazing areas than in the park suggests that the pastoral gazing areas are important for overall conservation of mammal biodiversity in the Tarangire ecosystem. This may because pastoral grazing areas provide an additional buffer to the Tarangire National Park and reduce effects of hard edges as suggested by Woodroffe and Ginsberg (1998).

These results also show that while anthropogenic activities may have negative impacts on species, the impact may be different depending on the intensity of human activities, and there are situations where man and wildlife can accommodate each other to the benefit of biodiversity conservation. The lower species richness especially large carnivores in the pastoral grazing areas shows that large carnivores are more sensitive to human activities which might be because of their biology i.e. they require large areas to survive but also often come into conflict with people outside protected areas for many reasons such as livestock predation and competitions for space and resources (Woodroffe and Ginsberg 1998). Whilst anthropogenic activities in pastoral grazing areas might have benefits for non-carnivore species richness, there were negative impacts on overall abundance. As discussed in chapter 3 the higher non-carnivore species richness found in pastoral grazing areas may be explained by the intermediate disturbance hypothesis which suggest that species richness tend to increase at intermediate levels of disturbance but decrease at higher or is lower at lower levels of disturbance (Connell 1978, Huston 1979). Previous studies suggest that pastoral rangeland management practices e.g. burning, tree cutting play an important role in influencing species richness in savannah habitats because they create a mosaics of habitats which facilitate species coexistence (Fairhead and Leach 1996, Nyerges 1996). Although non-carnivore species was higher in pastoral grazing areas, species relative abundances were generally lower in the pastoral grazing areas compared to the park. The lower relative abundance of species outside the park may be due to anthropogenic activities such as loss of wildlife habitats due to expansion of agriculture and legal and illegal wildlife hunting in pastoral grazing areas e.g. there were several occasions during the field work when I witnessed illegal hunters arrested by park rangers for hunting in pastoral grazing areas. Furthermore it also possible that these differences in species relative abundances between land use types is due to differences in ecological conditions. Tarangire National Park was established in order to provide protection of populations of migratory wildlife species during the dry season (Lamprey 1963, Lamprey 1964). The park contains a permanent water source – the Tarangire River which provides water to both migratory and resident wildlife populations during the dry season for the entire ecosystem (Prins 1987, Ishengoma et al. 2008). Therefore wildlife abundance is likely to be higher in the park because of the presence of permanent water source. Although some herbivores such as zebra and wildebeest move outside the park to the Simanjiro Plains during wet season and return during dry season (Kahurananga and Silkiluwasha 1997, Voeten 1999, Gereta et al. 2004), movement of mammals outside the park does not seem to explain the differences in mammal relative abundances between the park and pastoral grazing areas. This is because even during the wet season relative abundance of most mammals was higher in the park than in the pastoral grazing areas which may be because not all mammals move outside the park during the wet season.

Results also showed that relative abundance of most large bodied mammals decreased from the park through to grazing areas to cultivated areas. This is likely to be because large bodied mammals require intact and extensive habitats to survive (Sillero-Zubiri and Laurenson 2001, Sunquist and Sunquist 2001, Cardillo et al. 2004). However such habitats are becoming scarce because of increasing human population and demand for resources such as land for cultivation and human settlements. Furthermore large mammals are also the target of both legal and illegal hunters as they are considered to be most profitable (Caro et al. 1998, Fischer and Linsenmair 2001). The study therefore supports previous studies (Ottichilo et al. 2001, Serneels and Lambin 2001, Georgiadis et al. 2007, Stoner et al. 2007) which demonstrate that the conservation of large mammals outside protected areas is particularly challenging because of anthropogenic activities.

It can be concluded that although relative abundance of mammals showed a decreasing pattern with increasing anthropogenic pressure, human activities are unlikely to be the sole reason for this pattern. Other factors such as ecological differences between the park and areas outside the park are likely to have an influence e.g. naturally the park was established in an area that contains key habitats for maintaining wildlife populations in the ecosystem when compared to other areas such as the pastoral grazing areas in Simanjiro. However, given the importance of pastoral grazing areas in Simanjiro district, long term maintenance of wildlife population in the ecosystem is only possible if these areas are kept intact, but that can only be achieved through local community involvement in wildlife conservation which entails sharing of wildlife benefits and land use planning as discussed below.

7.3 Biodiversity in the Human Context

One of the most important challenges to conservation of biodiversity today is conflict of interest for land use in areas that are important for biodiversity, because they are also favoured by people and their activities often have negative impacts on biodiversity (Balmford et al. 2001a, Balmford et al. 2001b, Luck et al. 2004). This implies that it is unlikely that effective conservation of biodiversity can be achieved without the involvement of people, since people are part of many ecosystems and play an important role in shaping these ecosystems (Sanderson et al. 2002). Although ecological factors may be also important, this study has shown that mammal biodiversity decreased with increasing human pressure and therefore long term conservation of mammal biodiversity in the Tarangire ecosystem must involve local people outside the park. However it is important to understanding that local people's support for conservation may be influenced by their attitudes towards wildlife. In this section I discuss broadly the main findings on the attitudes of Maasai towards wildlife, especially large carnivores and suggest solution for resolving conflicts between Maasai and wildlife.

7.3.1 People's attitudes toward wildlife

In chapter 6 it was found that people generally held negative attitudes towards wildlife, especially large carnivores despite low level of livestock depredation by large carnivores. It was also shown that only few people received income from wildlife-related activities and the intensity of conflict increased with the level of wildlife knowledge, the density of problematic animals near human settlements and number of income sources. These findings have important implications for conservation of mammal biodiversity in this area. For example, the high level of conflict reported here, despite the low level of livestock losses to predators suggests that people do not support conservation in this region, not only because of the direct impact that wildlife has on their livestock, but it may also be because of other reasons which may include for example, competition between people and wildlife for important resources especially land. Previous study showed that there were disputes between management of the Tarangire National Park and villages in the area, with some villages such as Emboreet accusing the management of the park of attempting to extend the park boundary to village lands (Sachedina 2008). Such disputes are likely to affect negatively people's attitudes towards conservation in general.

It was interesting to find that intensity of conflict increased with wildlife knowledge and the number of income sources. Previous work elsewhere showed that relatively educated people (Infield 1988, Fiallo and Jacobson 1995) and those with diversified income sources (Infield 1988, Oli et al. 1994, Dickman 2008) were more tolerant towards wildlife conservation, probably because relatively educated people may see the importance of wildlife conservation and wealthy people are likely to cope better with wildlife-related losses such as livestock predation or crop damage (Naughton-Treves 1997, Cutter et al. 2000). The increase in the intensity of conflict with level of wildlife knowledge found here shows that people who knew certain wildlife species tended to have more negative attitudes than those who didn't. This is probably because those who were knowledgeable on wildlife species knew what were the potential problems from the species e.g. livestock predation and crop raiding and therefore were generally more negative. But it is also possible that people with knowledge of wildlife species were more knowledgeable on conservation needs of the species which in this case may also imply competitions for resources which increases hostility as discussed above. However further investigation will be important in order to ascertain the impact of local people's knowledge of wildlife species on their attitudes towards conservation.

7.3.2 Benefit sharing for resolving conflict between people and wildlife

In order to address the need for local people's support for conservation in the Tarangire ecosystem, Tanzania National Parks (TANAPA) through its Community Conservation Services, initiated community-based conservation in the Tarangire ecosystem, including the Simanjiro area. The aim of the program is to improve local community livelihoods and encourage support for conservation (Sachedina 2008, Sachedina and Trench 2009). However as with most community-based conservation initiatives, this initiative addresses problems at society level e.g. building village schools. Such benefits, although much needed in rural areas, are generally social rather than financial and cannot offset the cost of living with wildlife (Walpole and Thouless 2005). For example, in Kenya it was shown that farmers whose children had to remain home in order to protected crops from wildlife were not benefiting from community schools built by tourism (Walpole et al. 2003). Furthermore even where households receive dividends they are too little to offset individual costs borne by the people (Naughton-Treves 2001). The cost of living with wildlife is often borne by individuals and such cost may not be met through support directed at society level (Dixon and Sherman 1990, Balmford and Whitten 2003).

7.3.2.1 Extractive Use of Species

Extractive use of resources has been suggested as one way of resolving conflicts between local people and wildlife. This method is seen as being important because for many years people have benefited from direct extractive use of wildlife resources, and to date extractive use remains important for many people (Hutton 2003). It is assumed that in areas where extractive use of wildlife resources is important for local people's livelihood, then benefits of living with wildlife offset the costs which are therefore likely to be accepted by the people (Leader-Williams and Hutton 2005). There are two ways through which this may be implemented. One way is to allow the use of species that cause conflict with people or individual animals that cause conflict with people. It is suggested that by doing so there is a potential of reducing conflict through other means which may cause more problems such as lethal control, but also this method can provide meat to local people (Leader-Williams and Hutton 2005). However this method is probably most applicable if the hunted species is important for the diet of the people, such that with higher levels of off take it may reduce conflict (Leader-Williams and Hutton 2005). The second way of implementing extractive use of species is by allowing the use of species which do not cause conflict with people i.e. using natural resources other than problem species to offset conflict (Leader-Williams and Hutton 2005) e.g. extraction of non-timber forests products has been used to foster forest conservation and reduce conflict with local people in many areas in the tropics (Arnold and Peréz 2001).

Unfortunately extractive use has not been shown to be effective in improving people's tolerance to wildlife. The reason for the failure is that the extractive use approach was not designed to solve conflict between people and wildlife. In most cases it is used in combination with other measures that can provide benefits to local communities e.g. together with provision of conservation education, tourism and law enforcement (Leader-Williams and Hutton 2005). In Tanzania it was reported that despite the presence of regulations that require about 9% of game fees to be paid to local district councils (Leader-Williams et al. 1996), as a form of protected area outreach program (Barrow and Murphree 2001), the likelihood of the money reaching individual households who bear the cost of living with wildlife or poachers who use resources from Game Reserves was found to be small (Leader-Williams and Hutton 2005). Furthermore, in some cases extractive use has also been shown to have negative impact on species instead of resolving conflicts e.g. in tropical countries extractive use has led to overuse of wildlife species (Milner-Gulland and Mace 1998, Bennet and Robinson 2000).

7.3.2.2 Ecotourism

Ecotourism - tourism which is socially and more environmentally friendly (Giannecchini 1993). It is increasingly being used promote conservation because it enables local people to realise tangible benefits from wildlife in order to offset the costs of protecting wildlife and encourage coexistence (Goodwin 1996). Most wildlife tourism takes place in the protected areas but in recent year there has been an increasing trend in tourism development activities outside protected areas on private or communal lands e.g. Lewa Wildlife Conservancy in Kenya, Londolozi Reserve in South Africa and Save Valley Conservancy in Zimbabwe (Walpole and Thouless 2005). Investment in wildlife tourism in these areas has been reported to improve people's tolerance of wildlife (Frank et al. 2005). Although ecotourism appear to improve people's tolerance of wildlife, there are several limitations that may affect its effectiveness e.g. lack of access to capital. The initial costs may extremely high for local communities to afford. It requires a significant external investment that may not be recovered within a short period of time (Walpole and Thouless 2005). Lack of professionalism has also been pointed out as another limitation. It has been shown that where local communities have attempted to undertake their own tourism-related enterprises they have often failed due to lack of professionalism e.g. lack of attention on product quality, lack of understanding what tourist needs are and lack of appreciation of market realities (Roe et al. 2001). Where local communities have decided to enter into partnership with commercial operators in order to avoid these limitations, research suggest that they are likely to expose themselves to commercial risks which they do not understand and but also for which they have no control (Ashley and Roe 1998).

7.3.2.3 Wildlife Management Areas

For the Tarangire ecosystem, one possible solution for resolving conflict between people and wildlife would be through the development and establishment of WMAs as stipulated in the 1998 Wildlife Policy, where villages set aside land for the management of wildlife from which they can then benefit through wildlife related activities such as photographic tourism (URT 1998b). This approach provides tangible benefits to local communities which are clearly linked to wildlife resources, but also requires clear land use plans which are extremely important for the Tarangire ecosystem, given that some wildlife populations spend at least six months in the pastoral grazing areas in the Simanjiro Plains. Whilst such WMAs may be desirable in theory (see chapter 6), they can only be achieved if local communities are facilitated in the process by conservation institutions and the government, but also if the government addresses outstanding implementation problems. For instance, according to the WMA regulations, obtaining wildlife management and transferable user rights requires local communities to establish a multi-village level of associations that will be responsible for developing and implementing land-use and management plans, environmental impact assessments and a range of other requirements (Nelson 2008, Nelson et al. 2009, Trench et al. 2009). But, despite all this it is not clear when the government will relinquish control of hunting concession and income in WMAs in order to provide tangible benefits to local communities to have true ownership (Nelson 2008, Nelson et al. 2009, Trench et al. 2009).

CHAPTER 8: REFERENCES

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CHAPTER 9: APPENDICES

APPENDIX I: Semi-structured interviews used in this study

Questionnaire no:…………………..

1. Date	2. Survey no.	3. Individual ID	4. Interviewer
……./……./2007			

5. Household (GPS)	6. Village

7. Name	8. Age	9. Gender	10. Maternal language	11. When came to this village?	12. No of dogs owned

13. Since this time last year, has anyone in your family earned cash from/consumed: (Rank 1-3 in the table below).

	Yes	No	Rank (if needed)	Notes
Livestock?				
Crops?				
Off-farm work?				
Wildlife related				
Trophy hunting?				
Photographic tourism?				
Remittance?				
Other (specify)				

14. Who usually looks after your stock at this boma (e.g. laiyoni, morani, elders etc)?

Cattle ____
Smallstock ____

Chicken…………………………………………………………………………………………

Donkeys____

15. How are your stock at this boma tended to at night?

Cattle ____
Smallstock ____

Chicken………………………………………………………………………………………..

Donkeys____

16. Do you have a guard dog with your…… at this boma:

Cattle? Yes/No/NA **Smallstock?** Yes/No/NA **Donkeys?** Yes/No/NA **Chicken** Yes/No/NA

17. How many cattle/smallstock/donkeys/chicken do you have here?

Cattle	**Smallstock**	**Donkeys**	**Chicken**

<u>18. In the last moth, how many have ….</u>

	Sold ?	Slaughtered?	given away?	Stolen ?	killed by predators?	died from other causes?	Other losses? (specify)
Cattle							
Smallstock							
Donkeys							
chicken							

<u>Attitudes and knowledge</u>

19. Which of these animals (*show picture cards*) do you like/disklike in your fam and in grazing areas around your boma? Give reason (s).

Animal	Farm	Grazing areas	Like? /why?	Dislike?/why?	No opinion

20. Can you sort these pictures into animals that are a big problem, small problem or no problem around this boma, and explain why? *(show picture cards):*

	Identification		Problem?			Don't know animal	Doesn't occur here	Why is it a problem?
	Right Y/N	Spp confused with	Big	Small	No prob			
Giraffe								
Elephant								
Lion								
Leopard								
Cheetah								
African wild dog								
Wild cat								
Spotted hyaena								
Striped hyaena								
Impala								
Aardwolf								
Tiger								
Serval								
Caracal								
Buffalo								
Side striped jackal								
B-backed jackal								
Wildebeest								
Warthog								
Zebra								
Honey badger								
Zorilla								
Bat eared fox								
Slender mongoose								
Dwarf mongoose								
Egyptian mongoose								
White-tailed mongoose								
Banded mongoose								
African civet								
Common genet								
Large spotted genet								

APPENDIX II: Mammal photographs used to identify survey species

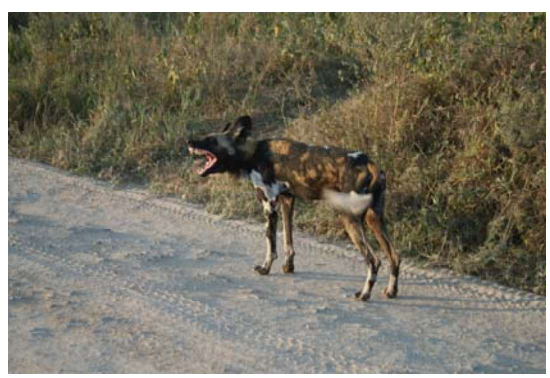
Giraffe

Elephant

Lion

Leopard

Cheetah



African wild dog

Wild cat

Spotted hyaena

Striped hyaena

Impala

Aardwolf

Tiger

Serval Caracal

Buffalo Side-striped jackal

Black-baked jackal Wildebeest

Warthog

Zebra

Honey badger

Zorilla

Bat-eared fox

Slender mongoose

Dwarf mongoose

Egyptian

White-tailed mongoose

Banded mongoose

African civet

Common genet

Large-spotted genet