LEOPARD POPULATION DENSITY,
HOME RANGE SIZE AND MOVEMENT PATTERNS
IN A MIXED LANDUSE AREA
OF THE MANGWE DISTRICT
OF ZIMBABWE

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ABSTRACT

Trophy hunting is often employed as a conservation management tool for large predators. However, in order for this method to succeed, hunting levels must be sustainable. Very little robust population data exist for African leopards (*Panthera pardus*) in general, and almost no density or spatial ecology data exist for leopards in Zimbabwe. Zimbabwe has one of the highest annual CITES leopard trophy hunting quotas in Africa, the sustainability of which has not been assessed, despite large scale landuse changes over the last 12 years. The focal area of this study was within the Mangwe district, in the south-west of Zimbabwe. The region is dominated by cattle and wildlife ranches, with high levels of leopard hunting, making it an important area for assessing leopard population density and spatial ecology.

Three population density estimation methods were employed in my study: a spoor index survey, an unbaited camera-trapping survey and a baited camera-trapping survey. Using three calibration equations, spoor indices appeared to underestimate the leopard population (1.28-3.29 leopards/ 100 km^2) as the equations were calibrated for areas with different habitats and leopard densities. In addition, the unbaited camera survey only produced six leopard photographs, unsuitable for individual identification and analysis. By contrast, the baited camera survey produced 292 identifiable leopard photographs, from which 13 individuals were identified. Density estimates calculated using the programme CAPTURE and the M_h model with the Mean Maximum Distance Moved Outside of Study Area (MMDMOSA) buffer method (4.79±0.83 leopards/100 km^2), and the programme SPACECAP, using a buffer of 2.5 km (5.12±0.62 leopards/100 km^2), appeared to generate the most reliable leopard population estimates. To assess the spatial ecology, three leopards (one male, two females) were captured and fitted with GPS collars. The home range estimates of the three leopards (95% Kernel UD: male 263 km^2, females 31 and 45 km^2) were smaller than those of leopards in more arid regions, but larger than those of mesic habitats. This suggests that the Mangwe area has a higher quality habitat than the arid regions of Namibia, but less suitable habitat than protected bushveld areas (e.g. Kruger National Park, South Africa).

My data represents the first robust leopard density and home range assessment for Zimbabwe. In addition, my results indicate that the current hunting quota issued to the Mangwe area is unsustainable. Consequently, I recommend revising the quota to five leopards for the entire area, and halving the current national leopard quota to 250, until a national leopard census is completed.

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LIST OF ACRONYMS AND ABBREVIATIONS USED WITHIN THE TEXT

AWT	- Africa Wildlife Tracking
CAR	- Central African Republic
CITES	- Convention on International Trade in Endangered Species of Wild Fauna and Flora
DRC	- Democratic Republic of Congo
GIS	- Geographic Information System
GPS	- Geographic Positioning System
HRT	- Home Range Tools
HMMDM	- Half the Mean Maximum Distance Moved
ICA	- Intensive Conservation Area
IUCN	- International Union for Conservation of Nature
LHZ	- Leopard Hunting Zones
MCP	- Minimum Convex Polygon
MCMC	- Markov-Chain Monte Carlo
MDM	- Maximum Distance Moved
MMDMOSA	- Mean Maximum Distance Moved Outside of the Study Area
NLMP	- National Leopard Management Programme
PAC	- Problem Animal Control
PDOP	- Positional Dilution of Precision
SECR	- Spatially-Explicit Capture-Recapture
SOAZ	- Safari Operators Association of Zimbabwe
UHF	- Ultra High Frequency
UTM	- Universal Transverse Mercator
VHF	- Very High Frequency
ZPWMA	- Zimbabwe Parks and Wildlife Management Authority

CHAPTER 1
GENERAL INTRODUCTION

"The embodiment of feline beauty, power, and strength" – Richard D Estes.

1.1 The leopard (*Panthera pardus*, Linnaeus 1758)

The leopard (*Panthera pardus*) has the widest and most diverse geographic distribution of any felid (Turnbull-Kemp 1967, Bothma & Walker 1999, Sunquist & Sunquist 2002). While the leopard is still broadly distributed across Africa and parts of Asia (Figure 1.1), its former range, which covered areas throughout northern Africa, sub-Saharan Africa, across south-east Asia, India, Sri Lanka, China, Tibet and the far east of Russia, has decreased (Sunquist & Sunquist 2002). It is estimated that, in Africa, leopards have disappeared from at least 37% of their historical range (Ray *et al.* 2005). The main reasons for this reduced distribution are attributed to habitat loss and retaliatory killing (Ray *et al.* 2005, Henschel *et al.* 2008) The most drastic range loss recorded to date has been from the Sahel belt, Nigeria and South Africa (Henschel *et al.* 2008).
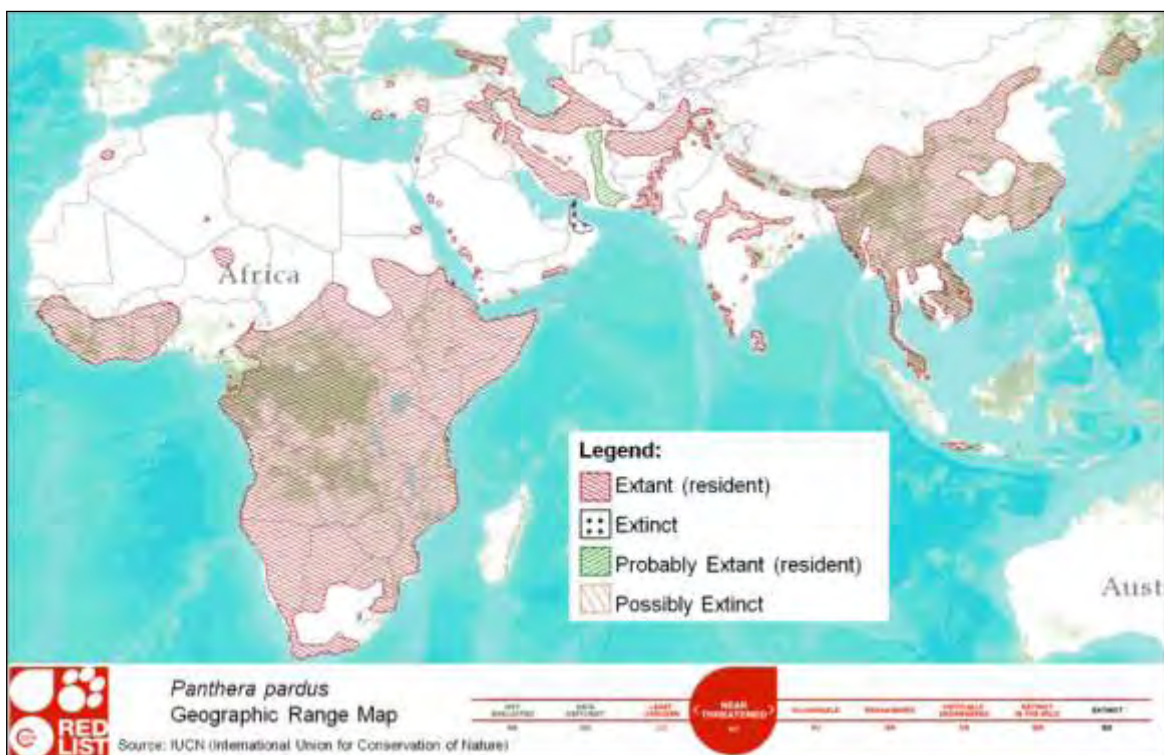
Figure 1.1: Current (2008) extent of the leopard's range (adapted from IUCN Red List 2008).

Nevertheless, the leopard's elusive and adaptive behaviour, and its extremely catholic diet have enabled it to survive across these different habitats, as well as near areas of dense human habitation, where other large cat species have long since been removed (Schaller 1973, Bothma & Walker 1999, Hayward *et al.* 2006). Although this widespread geographic distribution and adaptive behaviour offers little reason for concern for the survival of the species, these factors may be masking the more recent evidence that this secretive predator is disappearing from parts of its former range (Henschel & Ray 2003). Leopards are notoriously difficult to study because of their shy, solitary, and largely nocturnal behaviour. Consequently, very little robust population data exist (Bothma & Walker 1999). Also, leopards are not equally distributed throughout their range; different habitats will determine their varying spatial requirements, and therefore

distribution maps offer little information as to the densities or sustainability of certain leopard populations (Ray *et al*. 2005). Therefore, in some areas, leopard densities may be declining, and these trends and threats need to be investigated and monitored (Ray *et al*. 2005).

1.1.1 Conservation status

The conservation status of the leopard has been a matter of contention since the 1960s (Martin & de Meulenaer 1988, Nowell & Jackson 1996). The lucrative fur trade of the 1960s and 1970s reached such excesses that certain countries introduced bans on the export of felid skins (Loveridge *et al*. 2010). After 80 000 ocelot (*Leopardus pardalis*) and 15 000 jaguar (*Panthera onca*) skins were exported from Brazil in the mid-1960s, the country introduced a ban on wildcat skin exports in 1967 (Smith 1977). Between 1924 and 1960, 9162 leopard skins were exported from Uganda, and this led to the protection of the species there in 1960 (Loveridge *et al*. 2010). In 1968, 9556 leopard skins were imported to the United States of America (USA), followed by another7934 in 1969 (Martin & de Meulenaer 1988). It is estimated that the demand for leopard skins was approximately 50 000 skins per year in the early 1970s (Loveridge *et al*. 2010). This prompted a number of bans in the USA. In 1969 they introduced the 'Endangered Species Act of 1969' prohibiting the importation of live or dead animals, or parts and products thereof, for certain species listed in the Act (Martin & de Meulenaer 1988). In 1972, the leopard was added to this 'Endangered Species Act' which facilitated the collapse of the leopard fur trade in the USA (Martin & de Meulenaer 1988).

This wholesale exploitation of wildcat populations led to the first meeting for the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 1973, which became effective in 1975 (Martin & de Meulenaer 1988, Loveridge *et al*. 2010). The leopard was placed on CITES Appendix I, which includes species threatened with extinction, where the trade in these species is permitted only under exceptional circumstances (CITES 2011). In 1975, trade in leopard skins was not permitted by CITES, and trophy hunting lobbyists called for the leopard to be re-classified under CITES Appendix II as a threatened species, as opposed to its endangered classification under Appendix I (Martin & de Meulenaer 1988). The subsequent agreement to trade in leopard skins and trophies under CITES was granted to Botswana in 1978. Based on Hamilton's (1976) study of leopards in Kenya, leopards were then reclassified as threatened throughout most of sub-Saharan Africa, but remained on CITES Appendix I (Martin & de Meulenaer 1988).

After the fourth meeting of the Conference of Parties (CoP 4) of CITES in 1983, export quotas for leopard skins were granted to eleven other African countries (Martin & de Meulenaer 1988). This was as a result of the pressure exerted by these countries, who wanted to hunt leopard in

order to realise some form of economic benefit from their sale as trophies (Martin & de Meulenaer 1988). This trade agreement specified that a quota of leopards could be utilised in specific countries and that the products from these animals could be exported from those countries. However, the products could only be exported for personal use, not for trade or sale (CITES 2011), thus facilitating trophy hunting of leopards by allowing hunters to import their trophies to their country of origin (Hunter & Balme 2004). These national quotas were agreed upon by the CoPs, as they are today, as long as the export of these specimens does not endanger the survival of the species in the country of origin (CITES 2011). Except for Gabon and Kenya, the quotas for the African countries that are permitted to export leopard skins (Table 1.1), have either remained the same over the last 10 years or, as in the cases of Mozambique, Namibia, Tanzania and South Africa, been increased by at least twofold (CITES 2012). CITES increased these quotas based on arguments that the former quotas were inadequate to meet the demand, and therefore, the benefits of trophy hunting in these countries (Daly *et al.* 2005). It was also argued that, based on the growth of the game-farming industry in South Africa (and hence the increase in suitable habitat available to leopards) that leopard numbers have increased, and therefore, an increase in quotas was needed, although this was not based on any national population estimates (Daly *et al.* 2005).

Table 1.1: The annual CITES quotas allocated to the African countries permitted to export leopard trophies and/or skins* over the last 10 years (CITES 2012). Consecutive years where changes in quota allocations occurred are highlighted in grey.

Country	Quotas									
	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Botswana	130	130	130	130	130	130	130	130	130	130
DRC#*	-	5	5	5	5	5	5	5	5	5
CAR#	40	40	40	40	40	40	40	40	40	40
Gabon*	5	5	5	5	5	5	5	-	-	-
Ethiopia	500	500	500	500	500	500	500	500	500	500
Kenya*	80	80	80	80	80	80	80	80	80	-
Malawi	50	50	50	50	50	50	50	50	50	50
Mozambique	60	60	60	60	60	60	120	120	120	120
Namibia	100	100	100	250	250	250	250	250	250	250
Tanzania	250	500	500	500	500	500	500	500	500	500
Uganda	28	28	28	28	28	28	28	28	28	28
South Africa	75	75	75	150	150	150	150	150	150	150
Zambia	300	300	300	300	300	300	300	300	300	300
Zimbabwe	500	500	500	500	500	500	500	500	500	500
TOTAL	2118	2373	2373	2598	2598	2598	2658	2653	2653	2573

\# Democratic Republic of the Congo (DRC) & Central African Republic (CAR)
*Countries only permitted to export skins obtained from sources other than trophy hunting
  (eg. Problem Animal Control). No trophy hunting of leopards occurs in these countries.

These quotas are based on population estimates submitted by the countries party to the agreement. However, due to a lack of any alternative data, these estimates are based on Martin and de Meulenaer's (1988) overly simplistic, highly criticised, and outdated population estimation model (Norton 1990, Daly *et al*. 2005). This model was based on the relationship between leopard densities, and habitat type and rainfall, resulting in a regression equation between leopard density and rainfall (Martin & de Meulenaer 1988). Estimates for each country were then calculated based on the proportion of unmodified habitat in each country, which was assigned a mean annual rainfall in order to apply the regression equation (Martin & de Meulenaer 1988). This resulted in a total population estimate of 714 105 leopards for 41 African countries; with estimates of 23 472 and 16 064 leopards for South Africa and Zimbabwe, respectively (Martin & de Meulenaer 1988). The assumptions of Martin and de Meulenaer's (1988) model have been criticised as being too generalised, and the population estimates are thought to be unrealistically high, therefore recommendations for quotas based on these estimates should be viewed with caution (Norton 1990).

The International Union for Conservation of Nature (IUCN) Red List of Threatened Species bases threats to species on quantitative criteria, using thresholds for population size, range size, rate of decline, or probability of extinction (Macdonald *et al*. 2010a). The IUCN Red List threat categories, rated from least threatened to most threatened, are: Least Concern, Near Threatened, Vulnerable, Endangered, Critically Endangered, Extinct in the Wild, and Extinct. The leopard was demoted in terms of concern from Vulnerable in 1986, to Threatened in 1988, to Least Concern in 1996, and has recently been promoted in terms of concern to 'Near Threatened' since 2008 (Henschel *et al.* 2008). It has been suggested that due to the significance of the threats to the leopard in many parts of its distribution, it could soon qualify for the Vulnerable IUCN Red List category (Henschel *et al.* 2008).

1.1.2 Major threats to leopard populations
Habitat loss and natural prey depletion
Leopards are affected by both direct and indirect anthropogenic influences. The major threat to large carnivore populations, in general, is habitat loss and fragmentation resulting from the ever-expanding human population and the subsequent conversion of natural habitat to urban development and agricultural fields (Millennium Ecosystem Assessment 2005, Ray *et al*. 2005, Holmern *et al*. 2007, Loveridge *et al.* 2010). Habitat destruction and fragmentation through human activities such as logging and building of infrastructure (e.g. roads, mining activities and dams) have serious impacts on carnivore populations and their prey species (Loveridge *et al.* 2010). Carnivore populations are closely aligned to the abundance of prey species, and depletion

of this prey base can have a significant impact on carnivore population size (Karanth *et al.* 2004). Humans impact upon felid populations, albeit indirectly, by the depletion of natural prey species through over-hunting (Graham *et al.* 2005). Henschel (2007) reports how the bush-meat trade in the African forests has resulted in 'empty forest syndrome', where, although the forest habitats remain relatively intact, prey species have been decimated by over-hunting to the extent that the prey base cannot support viable leopard populations. This is becoming the case throughout much of Africa, where the socio-economic and political situations are promoting this human-predator conflict due to the competition for their shared limited natural resources (Graham *et al.* 2005).

Problem Animal Control (PAC)

Human-predator conflict is a major theme in carnivore conservation, and is especially pertinent to leopard conservation (Dickman 2010). Leopards are actively persecuted, legally and illegally, for the real or perceived threat that they pose to livestock and farmed game species, and occasionally to humans (Woodroffe *et al.* 2007, Balme *et al.* 2010b, Loveridge *et al.* 2010). There is no doubt that, in many instances, leopards are a real threat to livelihoods, mainly for small-scale livestock owners, whose loss of even a single animal will have a proportionally higher impact than that of larger-scale farmers (Loveridge *et al.* 2010). However, this retaliatory killing of leopards could pose a real threat to some local leopard populations. Historical evidence from the Western Cape in South Africa highlights this issue. Leopards were considered vermin up until 1968, even within national parks (Ray *et al.* 2005, Martins & Martins 2006); bounties were placed on them to encourage their eradication. This was so successful that, according to Hey (1964), leopards which were 'once common in all the mountainous areas of the Cape Province' have been 'eliminated from many areas and are becoming rare elsewhere'. Legal Problem Animal Control (PAC) accounts for a vast number of leopard deaths, probably equalling those resulting from trophy hunting (Balme 2009). However, illegal PAC by livestock farmers further compounds this issue as these deaths would be impossible to reliably quantify (Balme 2009). Any actions that can reduce livestock depredation are also likely to reduce the retaliatory killing of carnivores (Ogada *et al.* 2003), therefore, there is an obvious need to reduce livestock depredation by carnivores by improving livestock husbandry practices.

Hunting

Organised trophy hunting of wild animals can have considerable conservation benefits (Lindsey *et al.* 2007). This is especially important for developing countries which often lack the financial resources and infrastructure to adequately protect animals in national parks (Caro *et al.* 1998a). Large tracts of land are set aside for trophy hunting, an activity which generates a considerable

amount of revenue for governments, whilst affording some form of protection to wildlife through the prevention of human settlement encroachment to these areas (Caro *et al.* 1998a). It is also suggested that predators may be tolerated more favourably by land-owners if some form of compensation or revenue (e.g. through trophy fees) is generated (Balme *et al.* 2010a).

However, there is very little data investigating the impact that trophy hunting has on animal densities, and data that has been published by Balme *et al.* (2010a) from KwaZulu-Natal, South Africa, suggests that high hunting off-take can have a significant local effect on leopard populations. Mortality rates of the leopards in that study, especially of males, were more than double those recorded for leopards in similar habitats where no hunting occurred (Balme *et al.* 2010a). This over-exploitation of males could lower recruitment as incoming males tend to kill any cubs sired by the previously dominant male in the area (Bailey 1993, Balme *et al.* 2010a), and could also reduce the genetic diversity of the population by continually removing the fittest individuals as trophy animals (Harris *et al.* 2002). Therefore, hunting can often constitute additive, rather than compensatory mortality in leopard populations (Caro *et al.* 1998a, Balme *et al.* 2010a). This is further compounded by other human sources of off-take, such as PAC and poaching, which are not included in quota-setting calculations (Caro *et al.* 1998a, Balme *et al.* 2010a). Due to the difficulty in estimating carnivore populations, trophy hunting quotas are often based on guesswork (Caro *et al.* 2009), with no indication as to whether this projected off-take will be sustainable for the populations or not. Therefore, in order for hunting to be considered as a legitimate conservation tool, robust density estimates are needed, as well as monitoring programmes to ensure that off-takes are sustainable (Balme *et al.* 2010a).

Poaching for the skin trade and traditional medicines remains prevalent in many African countries, mainly for the domestic markets, but with some skins destined for international trafficking (Ray *et al.* 2005). In 1999, Künzel *et al.* (2000) counted 44 leopard skins in tourist shops in one day in Djibouti, and Shipp (2002) counted 17 skins within two hours in a market in Marrakech, and was offered another 20 verbally. In 2004, a skin trader in KwaZulu-Natal was arrested for being in possession of 58 leopard skins; it was alleged that these skins had been destined for regalia to be worn by the Shembe church and Zulu nobility (Carnie 2009). This trader was issued a suspended sentence, and was caught again in 2008 with another 64 leopard skins and he was still not prosecuted (Carnie 2009). This is a total of 122 leopard skins, which equates to the total hunting quota allocated to KwaZulu-Natal for the next 24 years. The quantification of illegal off-take is obviously difficult due to the nature of the practice, and any reports or estimates published are likely to be highly underestimated (Balme 2009). Therefore, in

order for trophy hunting and PAC quotas to be sustainable for leopard populations, decision makers will have to take these illegal practices into consideration.

1.2 The focus on Zimbabwe

1.2.1 Economically

Leopards are a prized trophy species, and are sold for a trophy fee of approximately US$4000 each, with a daily rate charged to the client of about US$1000 per day, and an average leopard hunt lasting 14 days (WM Grant, Professional Hunter, pers. comm.). This makes the leopard an important foreign currency earner for the country and the land owner (Grant 2008). In 1999, tourism accounted for 6.6% of the country's GDP, making it the second largest foreign-exchange earner after tobacco (ISS 2010). One of the main drawcards for tourism in Zimbabwe is the wildlife-rich national parks and safari areas, which make up about 16% of the country (Figure 1.2) (WWF-SARPO 2008, ISS 2010).
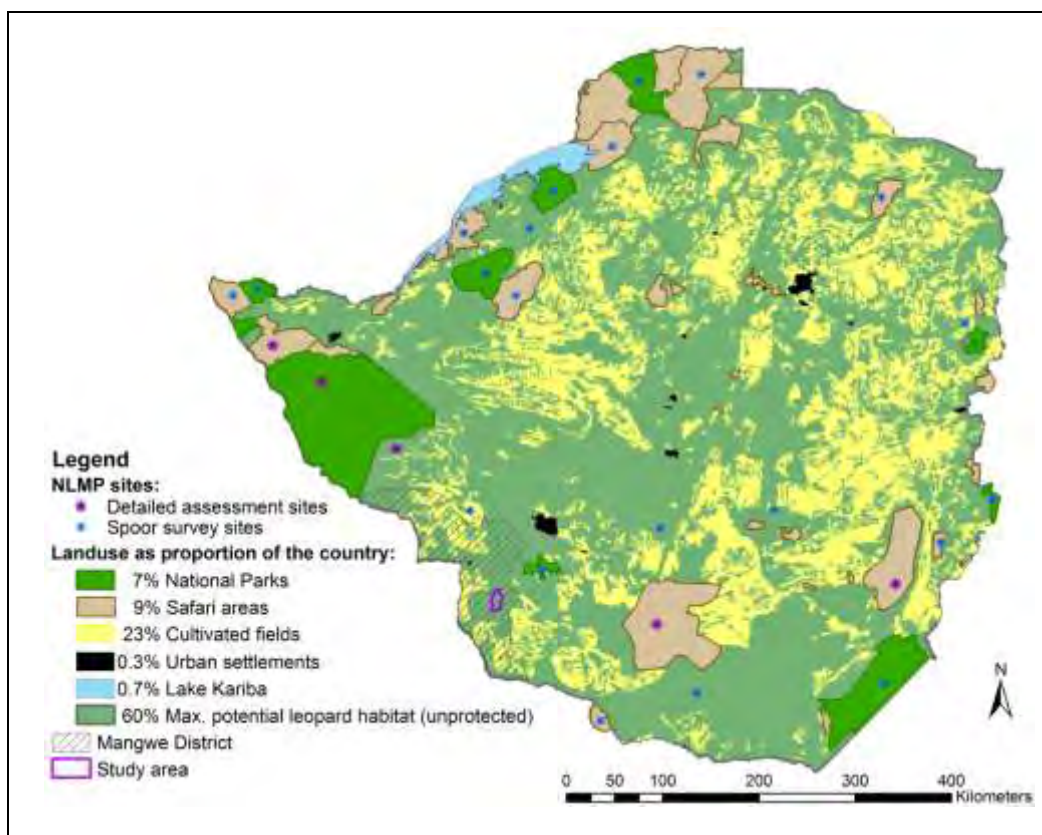
Figure 1.2: A map of Zimbabwe illustrating the proportions of different landuse types. The proportion of the country that could potentially support leopard populations with some form of protection (National Parks & Safari areas) is 16%, and the maximum potential habitat where leopards could occur, with no formal protection, is about 60% of the country. The Mangwe District and the focal study area of this thesis occur in the unprotected landuse area. The detailed assessment sites for the National Leopard Management Programme (NLMP) are indicated by the purple asterisks and the spoor survey sites for the NLMP by the blue asterisks.

1.2.2 Ecologically

Large carnivores, especially felids, have been revered as cultural icons and symbols throughout history. Their charisma draws the attention of the general public, and this has promoted their status as an umbrella, or flagship species, which has facilitated their conservation, as well as the conservation of their habitats, and the species within those habitats (Caro 2003, Loveridge *et al.* 2010). Large carnivores are wide-ranging, and therefore their effective conservation requires that vast areas of wilderness are preserved (Foreman 1993), although many of these protected areas are often insufficient in size for the survival of large carnivore populations (Linnell *et al.* 2001). Monitoring of large carnivore populations can alert biologists to changes in the ecosystem, as large carnivores are sensitive indicators of ecosystem change, even those at the lower trophic levels (Kucera & Zielinski 1995, Gros *et al.* 1996). Predators can alter primary production, through top-down processes, by regulating the size of herbivore populations (Soulé & Noss 1998, Terborgh *et al.* 2001). Apex predators determine the abundance of smaller mesopredators; if large carnivores are removed from the system, the numbers of mesopredators are likely to increase, increasing predation on the smaller prey species (Crooks & Soulé 1999). Leopards are the apex predator in the Mangwe District, after lions (*Panthera leo*) were extirpated in the early 1980s (WM Grant pers. comm.). Thus, their conservation is of utmost importance for the integrity of the ecosystem.

1.2.3 CITES

Zimbabwe has been party to, and has thus hunted leopard commercially, since the CITES agreement in 1983. The initial quota allocated to Zimbabwe was set at 80 leopards per year (Martin & de Meulenaer 1988). At the next CoP meeting in 1985, based on arguments by Child (1984) that the quota of 80 leopards per year was inadequate to meet the trophy hunting demand in Zimbabwe, the quota for the country was raised to 350 leopards, pending further research (Martin & de Meulenaer 1988). Based on Martin and de Meulenaer's (1988) results, Zimbabwe's annual CITES quota was then increased to 500 leopards in 1992 and has remained so to date (CITES 2011), even though the authors recommended that their population estimates were likely to decline by half in 20 years, due to a decrease in the availability of viable leopard habitat.

1.2.4 Politically

The focus on Zimbabwe is important because of the country's economic and political instability over the last decade as a result of the land reform process which started in 2000 (Purchase 2006, Child 2009). Not only did this land reform result in an increase in human encroachment on former wildlife areas, but it also resulted in poor conservation and wildlife management

practices, due mainly to the poor economic situation of the country (Purchase 2006, Child 2009). This lack of funds and capacity also impacted upon the resources available for wildlife research (Grant 2008). The Zimbabwe Parks and Wildlife Management Authority (ZPWMA) lacks the facilities and capacity to deal with the requirements needed for a rigorous monitoring system to determine whether quotas issued to certain areas are sustainable, which areas these quotas are actually being taken in, trends in trophy sizes, and the ratio of successful to unsuccessful hunts (Purchase 2006). Another issue that is not policed or reported on in Zimbabwe is the poaching of skins for commercial sale. These poached skins are often smuggled across the border into South Africa where the demand for them is high (Henschel & Ray 2003, Purchase 2006, Grant 2008).

1.2.5 Previous research

Very limited research has been completed on leopards in Zimbabwe and what little has been published is anecdotal and dated (Child & Savory 1964, Smithers 1966, Wilson & Child 1966, Grobler & Wilson 1972, Smith 1977). However, the results of Smith's (1977) study present the first home range data for leopards in Zimbabwe, suggesting that leopards in the Matopos area have relatively small home ranges in relation to leopards in the rest of southern Africa (Chapter 4: Table 4.1). Grobler and Wilson (1972) carried out a diet assessment of leopards in the Matopos National Park, where they found the leopards to be opportunistic in their choice of prey, showing a preference for small mammals, especially dassies (both *Procavia capensis* and *Heterohyrax brucei*) which were represented in 51% of all prey occurrences. The only recent data available for leopards in Zimbabwe is a brief assessment, commissioned by the ZPWMA, investigating the sustainability of off-take quotas in the Marula area, which was carried out by Purchase (2006). This report indicated that the off-take in the area had been greater than the estimated sustainable off-take (8% as per ZPWMA for leopard) since 2003, raising concerns about the viability of this leopard population. It was recommended that the national quota be reduced by at least 250, until a robust national leopard survey could be conducted (Purchase 2006).

1.3 Rationale for this study

Based on a combination of the above factors and the fact that Zimbabwe has had one of the highest annual CITES quotas (Table 1.1) since 1992, it is evident that an assessment of the sustainability of the current situation is needed. The collection of reliable and unbiased data is the essential component of such an assessment. In response to the alarming results of Purchase's (2006) report on the sustainability of trophy hunting in the Marula area, the ZPWMA responded by commissioning a National Leopard Management Programme (NLMP) in 2008. The NLMP was initially implemented by the Zambezi Society (ZamSoc) in collaboration with Oxford's Wildlife and Conservation Research Unit (WildCRU) and the ZPWMA. The project is currently being driven by the WildCRU as the 'Darwin Initiative for Biodiversity Leopard Project: Sustainable hunting of leopards in Zimbabwe', still in collaboration with ZamSoc and ZPWMA (NLMP 2008, WildCRU 2012). This NLMP aims to undertake a national leopard survey to assess the leopard populations in various habitats and landuse types throughout Zimbabwe. Detailed assessments are to be carried out in at least six different habitat and landuse areas (including the present study), as well as spoor count surveys in 20 different areas of the country (Figure 1.2) (NLMP 2008). Therefore, the present study will feed into this NLMP in order to provide the ZPWMA with an accurate assessment of the national leopard population in Zimbabwe.

The Mangwe area itself is an important representative site for the NLMP as it is not formally protected; it is a commercial livestock farming area which is subject to human encroachment, and it is impacted by poaching and trophy hunting (Grant 2008, NLMP 2008). This unprotected classification of landuse is likely to be the most common landuse type throughout the leopard's distribution in Zimbabwe (Figure 1.2) and it is therefore important to investigate leopard populations within it. According to the NLMP (2008), the hunting pressure in the Mangwe area is classified as 'high' consumptive use, with high levels of trophy hunting and PAC. Other areas that are being investigated are classified as 'non-consumptive' in the national parks to 'medium pressure' consumptive in hunting areas (NLMP 2008). Therefore, it is essential that the sustainability of off-take in this high hunting pressure area is investigated through population and home range estimates.

1.4 Aim and Objectives:

Aim

The aim of this study was to determine population density, and describe the spatial ecology of the leopards in the Mangwe area in south-western Zimbabwe, in order to provide reliable information for the future conservation and management of leopards in the area. This was achieved by addressing the following objectives:

Objectives

1. Determine the population density of leopards in the Mangwe area, using spoor tracking and camera-trapping techniques. Population density data are important for conservation and management decisions, quota-setting decisions in particular. These data will contribute to national leopard density estimates through the NLMP.

2. Determine the home range and movement patterns of leopards in the Mangwe area by using radio/GPS telemetry. Management decisions with respect to off-take (trophy hunting and problem animals) should be based on the biological boundaries determined by leopard home range sizes and movement patterns, and not administrative boundaries. Home range estimates will assist in determining the size of the areas that should be managed as an entity.

3. Recommendations for future management of the leopard in this area, based on the results of this study. These results will facilitate the development of a more informed and robust management plan for the leopards in this area, and the country as a whole. These data will feed into the larger NLMP.

1.5 Thesis structure

Chapter 1 - General Introduction: This chapter provides background information on the distribution and conservation status of the leopard, as well as the major threats to leopard populations. The status of leopards and leopard research in Zimbabwe is described, along with the motivation for the study.

Chapter 2 – Study Site: The study site is described in relation to the surrounding area and its history in the context of Zimbabwe.

Chapter 3 – Leopard Population Density Estimates: Density estimates are analysed using a spoor index survey and are presented based on calibrated results. Density estimates are also analysed using unbaited and baited camera-trapping methods, with various buffering techniques. The results of these estimates, using the programmes CAPTURE and SPACECAP are presented and discussed.

Chapter 4 – Leopard Home Range and Movement Patterns: The capture and collaring of three leopards is described. Home range, range overlap and movement patterns are assessed and described for the three collared leopards.

Chapter 5 – Synthesis and Management Recommendations: The thesis concludes with the key findings of the study and draws from the results obtained in the two data chapters to provide possible management recommendations for the future conservation of the leopard in the Mangwe area specifically, and in Zimbabwe in general.

CHAPTER 2

STUDY SITE
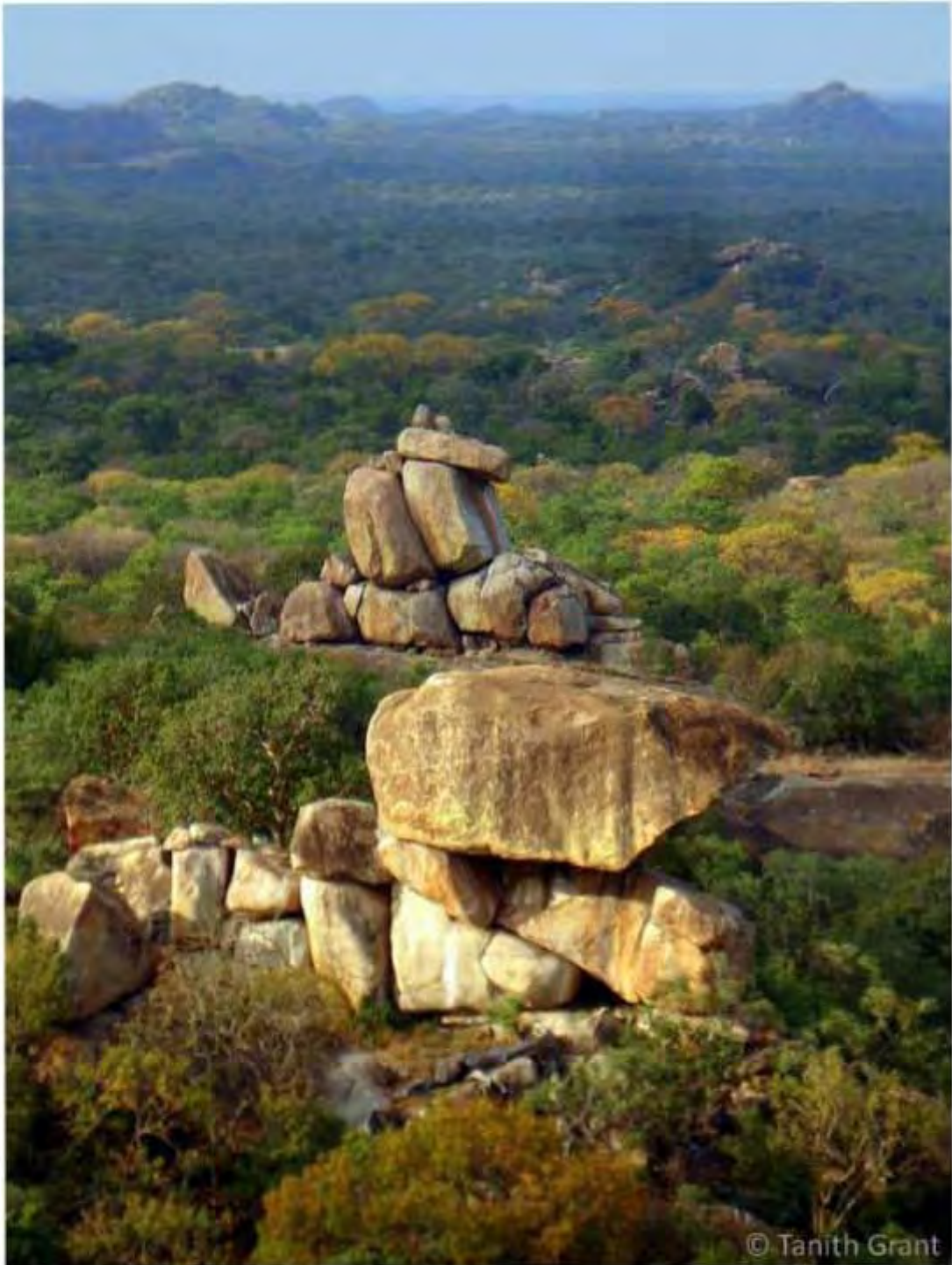
Typical 'castle kopjes', characteristic of the study area in the Mangwe District of Zimbabwe.

2.1 Introduction to study site

This study was conducted within a 200 km^2 section of the Mangwe District (Figure 2.1), formerly known as the BulilimaMangwe District, in south-western Zimbabwe. This district was characterised predominantly by commercial cattle (*Bos domesticus*) ranches until the land resettlement programme in 2000 (Purchase 2006). The area is now a mosaic of different landuses ranging from village resettlement/communal land, small single-owner cattle and/or game farms, to large commercial cattle and game farms. The study area is made up of seven small-scale, single-owner cattle farms, ranging in size from 16 to 38 km^2 (average 26 km^2) (Figure 2.1). These farms have been collectively managed as the 'Ingwezi Game Management Project' (Figure 2.1) since 2002 (Grant 2008). This 'Game Management Project' was initiated in response to the land reforms in the area, in order to conserve the game species on these small properties on a larger scale, to facilitate their consumptive use through trophy hunting (Grant 2008).

Unlike the Matopos National Park to the east (Figure 2.1), the Mangwe district is exposed to human encroachment, agriculture and hunting (for sport, poaching and personal use). These factors have been exaggerated since the dissolution of the Marula Intensive Conservation Area (ICA) in 2000 and the fragmentation of landuse practices as a result of the land resettlement programme (Purchase 2006). The Marula ICA incorporated the commercial farming properties (44%) and state-owned communal properties (56%) of the Marula and Mangwe areas from the mid-1980s, and the entire area functioned as a conservancy, managing the conservation of soils, water, vegetation and wildlife at a much larger scale than the individual farm boundaries (Purchase 2006, Metcalf & Sparrow 1998). The aim of the ICA was to optimise conservation and development objectives in the area, with an equitable cost and benefit allocation (Metcalf & Sparrow 1998). This ICA provided some form of wildlife protection through the sustainable consumptive use of game species by the wildlife industry (Grant 2008, Child 2009).

2.2 Climate

Zimbabwe is located in the sub-tropics, and the Mangwe District falls within the hot semi-arid climatic region (Peel *et al.* 2007). This region features precipitation and seasonal patterns like those of tropical savanna climates, with warmer wet seasons and cooler dry seasons (Peel *et al.* 2007). Zimbabwe uses a classification of agro-ecological zones for the various natural regions of the country and these are based on their potential for agricultural production, on the basis of soil quality, rainfall, vegetation and other factors (Ndebele *et al.* 2005, FAO 2006). The fertility of the region declines from Natural Region I to Natural Region V (Figure 2.1). The majority of the study area falls within Natural Region IV, with the southern section bordering (and slightly

overlapping) Natural Region V (Figure 2.1). These regions experience fairly low rainfall (< 650 mm per annum) and are thus suitable for semi-extensive to extensive cattle ranching or wildlife production, with little or no cropping (Ndebele *et al*. 2005, FAO 2006). Therefore, in the arid Natural Regions IV and V, the conservation of wildlife for consumptive purposes could be highly beneficial economically and ecologically, possibly even more so than livestock farming, if managed sustainably (Child 2009).

Zimbabwean Agro-ecological Regions:

Zone:		Farming System	Mean Annual Rainfall (mm)
	I	Specialised & diversified farming	>1000
	II	Intensive farming	750-1000
	III	Semi-intensive farming	650-800
	IV	Semi-extensive farming	450-650
	V	Extensive farming	<450
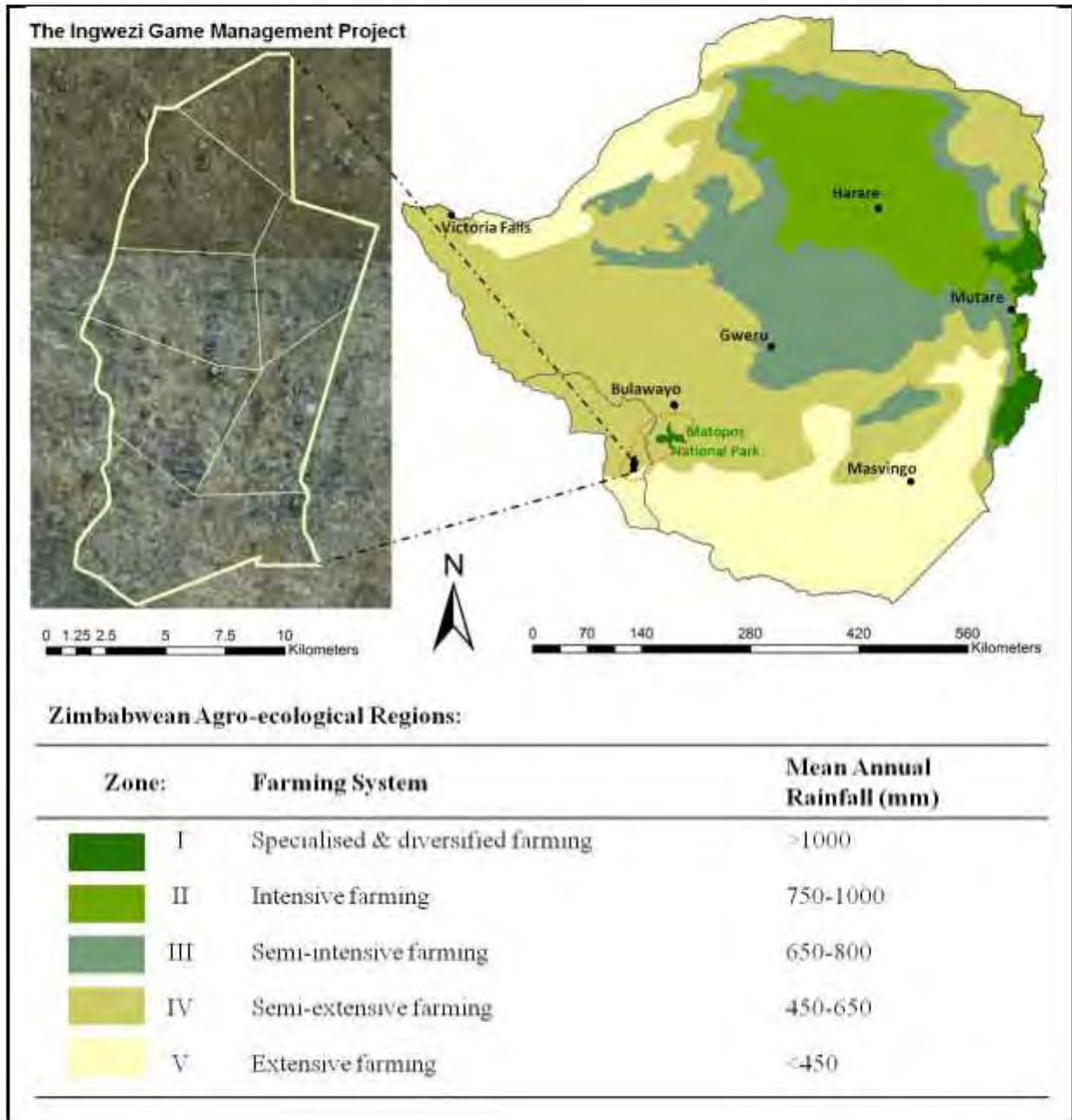
Figure 2.1: A map of Zimbabwe's agro-ecological regions, highlighting the location of the study area (black polygon) within the Mangwe District (grey outline) and the Matobo Hills range (dotted brown line). The Google Earth satellite image inset of the study area gives some indication of the density of the kopjes in the area and highlights the individual boundaries of the seven small-scale cattle ranches which are managed holistically as The Ingwezi Game Management Project.

2.2.1 Rainfall

The mean annual rainfall of the area is 615 mm (Tredgold 1956, Lightfoot 1981). This is usually concentrated within the six month period from October to March (Table 2.1), with December to February usually the wettest months and July to September the driest (Tredgold 1956).

Table 2.1: The 10-year average rainfall for three monthly periods for the Mangwe District (Tredgold 1956)

Three month period	Rainfall (mm)
July – September	3.6
October – December	236.5
January – March	327.2
April – June	30.7

2.2.2 Temperature

The mean annual temperature is 18.7°C, with the mean maximum temperature reaching 26.4°C, and a mean minimum temperature of 12.6°C. The hottest period of the year is during the months of September to November, with October normally being the hottest month. The mean monthly temperature for October is 20.9°C, with a mean maximum of 30.5°C and mean minimum of 14.4°C. June is normally the coldest month; the mean monthly temperature is around 12.3°C, with a mean maximum of 20.9°C and a mean minimum of 3.7°C (Tredgold 1956).

2.3 Topography and soils

The study area forms the western extent of the extensive Matobo hills range, made up of large balancing granite boulders known as kopjes which form dome-like outcrops (Barry & Mundy 1998). These kopjes are aligned into steep ridges and separated by deep valleys (Barry & Mundy 1998). These valleys and caves form places of refuge for both the leopards and their prey (Wilson 1969, WM Grant pers. comm.). The kopjes were formed by a rectangular system of jointing and subsequent erosion, resulting in the numerous 'castle kopjes', and hill slopes abundantly strewn with large boulders, and with some large bare granite whalebacks making up about 5% of the hill area (Lightfoot 1981).

Soils are derived predominantly from granite, and are immature because of the short distance from their site of formation to the site of removal, thus containing a high proportion of incompletely-weathered rock minerals (Lightfoot 1981). This has resulted in a fine sandy soil formed over deposits of Kalahari sands, with black clay deposits, and coarse-grained granitic sand (Moyo 2000). Profiles are rapidly leached of clay and fertility during the short but intense rainy season and, additionally, by run-off from the slopes of the kopjes. This makes the

woodlands very susceptible to erosion (Figure 2.2.c.) due to the poor soils and lack of grass cover (Lightfoot 1981).

Three main rivers and their tributaries dissect the study area: the Mangwe, Chavakadzi and Ntome Rivers. The rivers flow intermittently during the rainy season, but are dry for most of the year (Figure 2.2.f), with the exception of some small isolated pools. As in the Kalahari Gemsbok National Park of South Africa, these dry riverbeds and their riparian vegetation are important habitats for leopards (Bothma & le Riche 1984, Mills 1984). Leopards often navigate the landscape via these dry riverbeds, and the riparian vegetation is used for cover when hunting and for shade when resting during the day (Bothma & le Riche 1984).

2.4 Vegetation

The Mangwe area falls within the savanna (bushveld) biome (van Wyk & van Wyk 1997) and the overall vegetation type is described as deciduous tree savanna (Wild & Fernandes 1968). There are three general vegetation types in the area that are readily distinguishable – kopje vegetation, woodlands, and grasslands (Figure 2.2.e). On the kopjes, micro-climate and soil conditions change markedly over short distances; the semi-desert of the whalebacks support lichens (Figure 2.2.a & b) and the shallow pockets of soil on these kopjes support resurrection plant (*Myrothamnus flabellifolius*) and rock sedge (*Coleochloa setifera*), with occasional xerophytic trees (Figure 2.2.a &b) (Lightfoot 1981). Vegetation in the kopjes, between boulders, and at the base of kopjes, alters significantly, as plants benefit from the run-off from the grantie above and a build-up of soil and leaf matter (Figure 2.2.a & b) (Lightfoot 1981). These areas favour species such as mountain mahogany (*Entandophragma caudatum)*, paperbark corkwood (*Commiphora marlothii)*, and various *Euphorbia* and *Ficus* species.

Figure 2.2.a & b: Photographs of the study area illustrating the lichen growing on the edge of a whaleback (a), the vegetation supported between the boulders on the kopje (a & b), and some resurrection plants growing in shallow soil pockets (b).

Figure 2.2.c: Photograph of the study area illustrating the dominant *Colophospermum mopane* woodlands and the sandy soils which are easily eroded in the rainy season.

Figure 2.2.d: Photograph of the study area illustrating the second most dominant species in the area, *Terminalia sericea*.

Figure 2.2.e: A photograph illustrating the three general vegetation types in the area – kopje vegetation, woodland vegetation and the grasslands.

Figure 2.2.f: A photograph of one of the numerous dry river beds in the study area.

The woodlands host a mix of diverse species: *Colophospermum mopane* as the most dominant species, occurs throughout the area on the sodic soils (Figure 2.2.c)*,* followed by *Terminalia sericea* (Figure 2.2.d), and a mixture of other tree species such as *Vichellia, Kirkia*, *Commiphora*, *Euphorbia*, *Strychnos*, *Ficus*, *Combretum* and *Ziziphus*.

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The savanna grassland areas are dominated by perennial bunch grasses such as *Hyparrhenia filipendula, H. dissolute, Chloridion cameronii, Pogonarthria squarrosa* and *Heteropogon contortus* (Wild & Fernandes 1968).

2.5 Fauna

Lion (*Panthera leo*), cheetah (*Acinonyx jubatus*) and spotted hyaena (*Crocuta crocuta*) were extirpated from the area in the early 1980s, however the spotted hyaena started appearing again in about 2008 (WM Grant pers. comm.). Occasionally, lion and cheetah do pass through the area, however these events are rare, and the vagrants do not remain in the area long, as they are either chased off or killed by the famers (JR Peek, Biologist, pers. comm.). Hence, the leopard is the apex predator in the area, with brown hyaena (*Parahyaena brunnea)* the only other large carnivore commonly found in the area (pers. obs.). No published data exists on the fauna of the area, therefore a species list has been compiled based on the personal observations of a resident wildlife biologist (JR Peek pers. comm.), those of an experienced professional hunter who has hunted in the area for over 30 years (WM Grant pers. comm.) and based on the results of the camera-trapping surveys I conducted in the area and my own personal observations whist conducting the fieldwork for this study (Table 2.2). Based on the camera-trapping data, 30 mammal species, 11 bird species and one reptile species were identified (Appendix II).

The top mammalian prey species of the leopard, according to Grobler & Wilson's (1972) diet study in the nearby Matopos National Park, are common in the study area, and include yellow-spotted and rock hyraxes, impala (*Aepyceros melampus)*, klipspringer (*Oretragus oretragus)*, scrub hare (*Lepus saxitilis*) and duiker (*Sylvicapra grimmia)* (Table 2.2).

Table 2.2: A list of species likely to occur in the area based on the personal observations of resident biologist Richard Peek, the personal observations of local professional hunter Wayne Grant and of myself, including data from the camera-trapping surveys. **C**-Common, **FC**-Fairly Common, **R**-Rare, **VR**-Very Rare, **V**-Vagrant. These categories are based on those of Grobler & Wilson (1972) whose observations from the Matopos National Park have been included here for comparative purposes.

Family	Species	Common name	Grobler & Wilson (1972)	JRPeek pers. comm.	WMGrant pers. comm.
Canidae					
	Canis mesomelas	Black-backed jackal	FC	C	C
	Canis adustus	Side-striped jackal	V	VR	VR
Bovidae					
	Tragelaphus oryx	Eland	FC	FC	R
	Tragelaphus strepsiceros	Greater kudu	FC	FC	FC
	Aepyceros melampus	Impala	C	C	C
	Connochaetes taurinus	Blue wildebeest	C	C	C
	Oretragus oretragus	Klipspringer	C	C	C
	Raphicerus campestris	Steenbok	C	C	FC
	Sylvicapra grimmia	Common (grey) duiker	C	C	C

Cercopithecidae

	Cercopithecus pygerythrus	Vervet monkey	C	C	C
	Papio hamadryas	Chacma baboon	C	C	C

Equidae

	Equus burchelli	Burchell's zebra	R	FC	C

Felidae

	Caracal caracal	Caracal	VR	R	R
	Felis lybica	African wildcat	R	FC	FC
	Leptailurus serval	Serval	C	R	R
	Panthera pardus	Leopard	FC	FC	FC
	Panthera leo	Lion	V	V	V
	Acinonyx jubatus	Cheetah	-	V	V

Galagidae

	Galago moholi	Lesser bushbaby	-	FC	C

Herpestidae

	Galerella sanguinea	Slender mongoose	C	FC	FC
	Mungos mungo	Banded mongoose	-	FC	FC
	Ichneumia albicauda	White-tailed mongoose	R	R	R
	Helogale parvula	Dwarf mongoose	C	FC	FC
	Rhynchogale melleri	Meller's mongoose	R	R	VR
	Atilax paludinosus	Water (marsh) mongoose	C	FC	R
	Paracynictis selous	Selous's mongoose	FC	VR	R

Hyaenidae

	Crocuta crocuta	Spotted hyaena	-	R	FC
	Parahyaena brunnea	Brown hyaena	V	C	C
	Proteles cristatus	Aardwolf	R	VR	R

Hystricidae

	Hystrix africaeaustralis	Porcupine	FC	C	C

Leporidae

	Lepus saxitilis	Scrub hare	C	C	C
	Pronolagus randensis	Jameson's red rock rabbit		R	FC

Mustelidae

	Mellivora capensis	Honey badger	R	FC	C
	Aonyx capensis	Cape clawless otter	C	FC	C
	Ictonyx striatus	Striped polecat (zorilla)	-	VR	R

Orycteropodidae

	Orycteropus afer	Antbear (aardvark)	C	C	C

Pedetidae

	Pedetes capensis	Springhare	C	C	C

Procaviidae

	Heterohyrax brucei	Yellow-spotted hyrax (dassie)	C	C	C
	Procavia capensis	Rock hyrax (dassie)	C	C	C

Rhinolophidae

	Rhinolophus sp.	Horseshoe bat		C	C

Sciuridae

	Paraxerus cepapi	Tree squirrel	C	C	C

Suidae

	Potamochoerus porcus	Bushpig	C	C	C
	Phacochoerus africanus	Warthog	C	R	R

Viverridae

	Civettictis civetta	African civet	R	FC	R
	Genetta tigrina	Rusty spotted genet	C	C	C

2.6 Landuse and ownership

Land ownership has been a matter of contention since the early 1980s, and even more so after the Land Resettlement Programme of 2000 (Grant 2008). Figure 2.3 shows the landuse changes from the previous large commercial farms in the Marula ICA to either villages/communal land or new smaller commercial farms after the year 2000. To date, more landuse changes have taken place, but these are not reflected here because accurate data were unavailable.
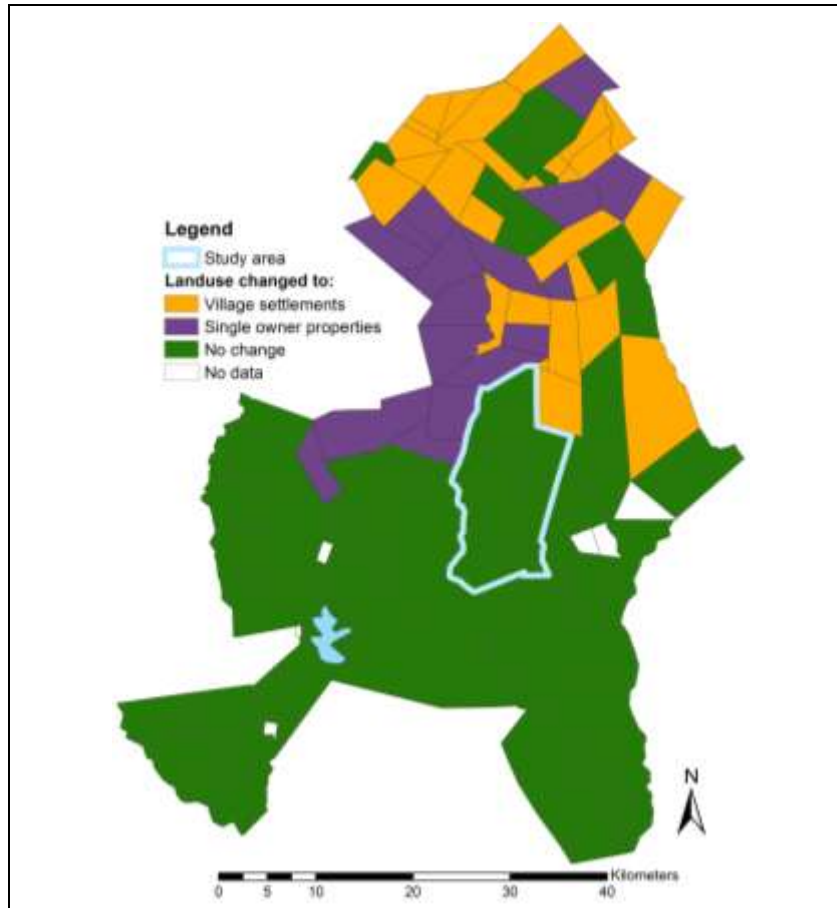
Figure 2.3: The former Marula ICA area showing the changes in landuse after the resettlement programme, post 2002 (adapted from Purchase 2006). The former landuse types (green) have not been specifically defined, nor have any changes in landuse post 2006, due to lack of available data.

A summarised chronology of the landuse in this area is as follows:

- Until the mid-1980s the majority of land occupation in the area was made up of 'large' commercial cattle farms (farms with private title deeds and wildlife use rights) referred to under the Revised Parks and Wildlife Act of 1996 as 'alienated land' (Metcalf & Sparrow 1998, Purchase 2006). These were single-owned, large (45 km^2), private ranches (Metcalf & Sparrow 1998)

- From about 1983, some of these commercial farms were sold to the new Zimbabwean government on a willing-seller, willing-buyer scheme. This scheme was devised under the British-brokered peace agreement of 1980, whereby Britain provided funds to the new

government to purchase any agricultural land that was offered to them, but they could not force farmers to sell their land (Grant 2008). This resulted in 56% of the commercially farmed area being handed over to the government, which it then allocated to the Agricultural Rural Development Authority (ARDA) for commercial beef production. The remaining 44% was retained by the 'large' commercial farms (Metcalf & Sparrow 1998).

- The ARDA programme did not thrive, and in 1988 new land classifications were devised for the ARDA areas. The government advertised for potential farmers under the objective of Black Economic Empowerment (BEE) to apply for these 'new farms'. This resulted in the following two land classifications:

    - 24% of the area was allocated as grazing access right, whereby no human settlement was allowed, but grazing access was granted to neighbouring communal land residents (Metcalf & Sparrow 1998).

    - 32% of the area was allocated as the 'new commercial settlement farms'. This was made up of 35 smaller scale units of land intended for cattle production, each between 15-20km^2 (Metcalf & Sparrow 1998).

- The 44% still retained by the 'large' commercial farms was subjected to a compulsory designation exercise which was only addressed again in the year 2000 (Metcalf & Sparrow 1998).

- The year 2000 saw the realisation of the proposed land resettlement programme mentioned in the late 1980s and early 1990s. This is still an ongoing process and has resulted in about 22% of the 'large' commercial farms being redistributed without any compensation to the original land owners (Grant 2008).

This land reform has resulted in the fragmentation of what used to be large, single-owned properties which were managed collectively as the Marula ICA. The new, smaller-scale farms can have one family residing on the land or multiple owners, with up to 100 families residing on the same properties (Grant 2008). This has resulted in a dramatic increase in the number of people and livestock in the area, reducing the habitat available for wildlife (Purchase 2006, NLMP 2008). It is evident that the reintroduction of an ICA or some form of co-ordinated large scale wildlife management is needed in the area. These relatively small farm units and fragmented land should be incorporated into a larger-scale conservation management and monitoring programme, which has the potential to benefit all involved if the costs and benefits are equally distributed (Metcalf & Sparrow 1998).

CHAPTER 3

LEOPARD POPULATION DENSITY ESTIMATES

Camera-trap photograph of one of the collared female leopards (F1) captured during the baited camera-trapping survey.

3.1 Introduction

Robust abundance and population density estimates are key baseline parameters in any wildlife or conservation management project (Stander 1998, Gusset & Burgener 2005). Without this baseline information, conservation management decisions are often based on crude estimates, which could result in misguided decisions that can be counter-productive, or even detrimental for conservation (Blake & Hedges 2004, Jhala *et al.* 2011). With most large carnivore populations declining worldwide (Nowell & Jackson 1996), practical and accurate methods of estimating populations and monitoring demographic trends should be a priority for conservation management (Stander 1998). Animal abundance is assessed in two ways, either as relative abundance, or absolute abundance (Gese 2001). Relative abundance is usually measured through indirect methods which use indices of animal abundance that can be compared temporally and spatially (Henschel & Ray 2003). Absolute abundance involves more direct methods that actually count animals and then estimate population density (Gese 2001).

Estimating animal numbers is often practically difficult, time consuming and expensive. Cryptic carnivores are secretive in nature, generally nocturnal, dangerous in demeanour and occur at low densities, making them especially difficult to sample using direct methods (Gros *et al*. 1996). Consequently, several indirect techniques for measuring density are usually employed, but they must be cost-effective, repeatable and objective (Smallwood & Fitzhugh 1995, Stander 1998, Gusset & Burgener 2005). Such indirect approaches include public participation through interviews, or the public are asked to provide photographs from which a variety of data can be extracted (Maddock & Mills 1993). Alternatively, predator densities are modelled based on habitat quality and prey biomass (Smith 1977, Martin & de Meulenaer 1988, Mladenoff *et al*. 1995). Presence of animal signs is a popular indirect sampling method. This can be through scent stations (Harrison 2006, Barea-Azcón *et al.* 2007), transects in order to detect presence of scat, scrapes, scent markings or spoor (McCarthy *et al.* 2008), vocalisation response surveys (Maddock *et al.* 1996, Ogutu & Dublin 1998), or line-transect sampling where animals encountered are counted, and inferences are made about the population through various indices (Smallwood & Fitzhugh 1995, Voss *et al.* 2001, Silveira *et al*. 2003, Thoisy *et al*. 2008). These methods have their advantages; however, some of the requirements and assumptions of the methods make them less robust than a direct approach. Indices as a measure of abundance can be fraught with potential error and bias (Anderson 2001, McCarthy *et al.* 2008). Assuming that detection probability is constant across all factors, such as habitat types, observers and different weather patterns can be problematic (Anderson 2001). On the other hand, direct censuses or complete counts of animals are often impractical, expensive and time-consuming (Balme *et al.* 2009a). They are also based on several assumptions and frequently use estimators, as it is almost

impossible to count every individual with certainty in a free-roaming population (Henschel & Ray 2003). Capture-mark-recapture sampling and telemetry are two of the most commonly used direct sampling methods.

Spoor indices have frequently been used to measure the relative abundance of carnivores (Van Dyke *et al*. 1986, Smallwood & Fitzhugh 1995, Stander 1998, Hayward *et al*. 2002, Gusset & Burgener 2005, Davidson 2009, Houser *et al*. 2009, Funston *et al*. 2010). Spoor indices are often less invasive and more cost effective than direct methods; they are also repeatable and provide a measure of precision for any estimate (Houser *et al*. 2009, Funston *et al.* 2010). Initial spoor surveys carried out by Stander (1998) relied on the highly specialised and accurate tracking skills of experienced San hunters, whereby individual leopards were identified by their spoor. Balme *et al*. (2009a) were also able to identify individual leopards by their spoor. This was facilitated by radio-collaring all known leopards, except for one, in the study area and using these telemetry data to verify the track data (Balme *et al*. 2009a). Except for these very exceptional applications of the method, broader application of the method is limited, as most tracking studies do not have the resources required to identify individuals from their spoor with any certainty, violating many of the underlying assumptions of the method (Balme *et al*. 2009a). Another factor influencing the accuracy of spoor-tracking, is the substrate where transects are conducted. It may be more difficult to discern tracks in harder substrates (such as clay) than softer ones (such as sandy soils) (Stander 1998, Henschel & Ray 2003, Funston *et al*. 2010). Therefore, in most spoor-tracking methodologies, only the species' tracks and their sex can be identified (Houser *et al*. 2009, Funston *et al*. 2010). Abundance is therefore estimated by calculating spoor indices, which require calibration with true densities (Stander 1998, Gusset & Burgener 2005, Funston *et al*. 2010). However, true densities are not always available for different study areas or terrains and they are often logistically and financially prohibitive, and extrapolation of existing calibrations across areas and species is problematic (Balme *et al*. 2009a, Davidson 2009).

Camera-trapping has been successfully employed as a theoretically sound and practically feasible method for estimating the population densities of cryptic carnivore species (Karanth & Nichols 1998). The individual identification of certain species is facilitated by the stripe, spot or rosette patterns which are as individual to each animal as a fingerprint is to a human (Miththapala *et al*. 1989, Henschel & Ray 2003). Identification of individuals allows one to utilise mark-recapture, or capture-recapture surveys to estimate population densities in certain areas (Otis *et al*. 1978, White *et al*. 1982). This method has been used for numerous species across the world; tigers (*Panthera tigris*) in India and Malaysia (Karanth 1995, Karanth & Nichols 1998, O'Brien *et al.* 2003, Kawanishi & Sunquist 2004, Karanth *et al.* 2006, Royle *et al.*

2009, Wang & Macdonald 2009), jaguars in Central and South America (Maffei *et al.* 2004, Silver *et al.* 2004, Soisalo & Cavalcanti 2006), ocelots in Brazil (Trolle & Kery 2003) and Belize (Dillon & Kelly 2008), snow leopards (*Uncia uncia*) in India (Jackson *et al.* 2006), cheetahs in South Africa (Marnewick *et al.* 2008), brown hyaenas in South Africa (Thorn *et al.* 2009) and leopards throughout their range ((Henschel & Ray 2003 (Gabon), Steyn 2007 (Botswana), Edgaonkar 2008 (India), Khorozyan 2008 (Iran), Stein *et al.* 2008 (Namibia), Wang & Macdonald 2009 (Bhutan), Balme *et al.* 2009a (South Africa)).

Capture-recapture analysis relies on three fundamental assumptions that need to be satisfied during any camera survey (Otis *et al.* 1978, White *et al.* 1982). The first is that the population sampled should be demographically and geographically closed, meaning that there should be no births, deaths, immigration or emigration for the duration of the sampling (White *et al.* 1982). In order to satisfy this assumption, camera-trapping surveys are generally carried out over a relatively short period of time, usually no longer than three months (Karanth & Nichols 1998, Silver *et al.* 2004, Jackson *et al.* 2005). The second assumption is that all individuals have an equal chance of being captured, meaning that no animal has zero-capture probability (Karanth & Nichols 1998). Therefore, trap sites and trapping effort should be equal across the study area, without any gaps within which an individual could range without being captured (Karanth & Nichols 1998). Finally, all individuals have to be reliably distinguishable from each other, throughout the duration of the study, to determine whether they have been captured or recaptured (White *et al.* 1982, Silver *et al.* 2004, Jackson *et al.* 2005). Hence, camera placement and image quality are of utmost importance when attempting to identify individual leopards by their distinct rosette patterns (Jackson *et al.* 2005).

Leopard density estimates have been calculated throughout the species' range. Table 3.1 summarises all the available leopard density estimates using camera-trapping and spoor-tracking techniques. Leopard densities vary depending on habitat type (Table 3.1). Leopards found in more arid regions, such as Namibia and Iran, tend to have lower densities than those found in more mesic environments, such as KwaZulu-Natal (South Africa) and India (Table 3.1). Inter-specific competition will also influence leopard densities. For example, Edgaonkar (2008) found that leopard densities were perceptively higher in the Sariska Reserve where tigers had been extirpated (Table 3.1). Other factors affecting leopard densities include habitat alteration, prey availability (Mizutani & Jewell 1998, Treves & Karanth 2003) and the degree of protection afforded in certain areas (Caro *et al.* 1998a, Marker & Dickman 2005a, Balme *et al.* 2009b). Productive leopard habitat may have low leopard densities if leopards are persecuted either through trophy hunting, poaching or habitat loss due to human encroachment (Caro *et al.* 1998b,

Treves & Karanth 2003, Balme *et al.* 2010a). However, caution must be exercised when comparing these densities as the methods involved in estimating the densities may not be directly comparable.

Table 3.1: Summary of literature on leopard density estimates using spoor count indices and camera-trapping methods.

Authors	Location	Habitat	Method	Buffer method	Density (Leopards/ 100 km^2)
Balakrishnan & Easa 1986	Parambikulam Wildlife Sanctuary, Kerala, India	Wet evergreen forest, tropical semi-evergreen forest, secondary moist mixed deciduous forest	Spoor-tracking		2.13
Balme *et al.* 2009a	Phinda Private Game Reserve, KwaZulu-Natal, South Africa	Lowveld bushveld and bushveld-grassland	Spoor-tracking	HMMDM* (Capture-recapture)	6.45
Smith 1977	Rhodes Matopos National Park, Rhodesia	Open woodland and grassland	Spoor-tracking		17
Stander 1998	Tsumkwe District Communal Area, Namibia	Semi-arid savanna woodland	Spoor-tracking & telemetry		1.45
Balme *et al.* 2009a	Phinda Private Game Reserve, KwaZulu-Natal, South Africa	Lowveld bushveld, coastal, and bushveld-grassland	Camera-trapping	MMDMOSA* HMMDM [1] HMMDM [2]	5.19 4.99[1] 3.29[2]
Chapman & Balme 2010	Zululand Rhino Reserve, Kwa-Zulu Natal	Valley lowveld: bushveld and open savanna thornveld	Camera-trapping	HMMDM (with CAPTURE models M_o & M_h)	M_o: 2.5 M_h: 7
Chauhan *et al.* 2005	Sariska Tiger Reserve, India	Dry forests	Camera-trapping		23.5
Edgaonkar 2008	Satpura Tiger Reserve & Sariska Tiger Reserve, India	Moist decidous forest (Satpura) and tropical dry decidous and thorn forest (Sariska)	Camera-trapping	HMMDM MMDM	Satpura: 8.02, 5.07 Sariska: 30.9, 20.7
Henschel 2008	Lopé and Ivindo National Parks, Gabon	Lowland forest	Camera-trapping		2.7 – 12.1
Khorozyan 2008 (Persian leopard *Panthera pardus saxicolor*)	Bamu National Park, Fars Province, Iran	Semi-arid temperate	Camera-trapping	N/A	1.9
Stein *et al.* 2011	Waterberg Plateau Park and Farmlands, north central Namibia	Thick shrubland and woodland	Camera-trapping	MHHR*-from telemetry	Park: 1.0 Farm-land: 3.6

| Steyn 2007 | Northern Tuli Game Reserve, Botswana | Low-tree savanna and shrub thicket | Camera-trapping | HMMDM MHRR HMHRR* | 10.2 7.5 4.6 |
| Wang & Macdonald 2009 | Jigme Singye Wangchuck National Park, Bhutan | Sub-tropical plains to alpine forest | Camera-trapping | HMMDM | 1.04 |

*HMMDM – Half the Mean Maximum Distance Moved
*MMDMOSA – Mean Maximum Distance Moved Outside the Study Area
*MHHR- Mean Home Range Radius
*HMHRR – Half the Mean Home Range Radius
[1] Estimated from camera-trap data
[2] Estimated from telemetry data

Since Smith's (1977) paper on leopard movement patterns, based on spoor-tracking in the Matopos National Park, there has not been any published research on leopard densities in Zimbabwe (Table 3.1). Significantly, this estimate is not considered representative for the country as a whole because the Matopos National Park is recognised as having a higher leopard density than other areas (Smith 1977, ICOMOS 2003). This is thought to be due to the high density of kopjes in the area, which provide high-quality habitat for leopards and their prey, and because the area is a national park (ICOMOS 2003). Leopard off-take permits in Zimbabwe continue to be issued based on the exaggerated and outdated density estimates of Martin & de Meulenaer (1988) (see Chapter 1). This highlights the need for robust and repeatable leopard density estimates in Zimbabwe (Norton 1990). The Mangwe area, in particular, is an important site to estimate leopard densities as it is a non-protected area, and it is impacted by trophy hunting, poaching and cattle farming. Importantly, this unprotected, human-impacted landuse type represents the majority of the leopard's habitat throughout its range (Nowell & Jackson 1996). In Zimbabwe, up to 60% of the country could support leopards outside formally protected areas (see Chapter 1: Figure 1.2). Thus the development of a robust estimate and approach for determining leopard densities in the Mangwe area will have important implications for leopard conservation and sustainable off-take quotas for Zimbabwe as a whole.

3.2 Objectives

The objective of this chapter was to accurately estimate the density of leopards in the 200-km^2 study area using both indirect and direct approaches (spoor-tracking transects and camera-trapping capture-recapture methods).

3.3 Methods

Spoor-tracking was conducted from 16 August 2010 to 1 September 2010. Two camera-trapping surveys were conducted, the first was an unbaited survey conducted from 23 October 2009 for 40 days and the second was a baited survey from 26 June 2010 for 66 days.

3.3.1 Spoor index survey

All the roads in the study area were driven and assessed for suitability in terms of soil type, direction, habitat, length and traffic (Stander 1998). A handheld GPS (Garmin 72) was used to log the tracks and these paths were entered into ArcGIS 9.3.1 (ESRI Inc., California) to measure and select appropriate transects. The number of transects required and their length should be determined by the size of the study area, in order to achieve sufficient sampling effort (Stander 1998, Balme *et al*. 2009a, Houser *et al*. 2009). This relates to an index known as the penetration rate and is defined as the sum of the combined road length of all transects, expressed as a ratio of 1 km of road to *x* km^2 surface area of the study area (Stander 1998). Thus, a high effort of penetration will be reflected by a low number (Houser *et al.* 2009). There were limited roads within the study area where leopard tracks could be detected. Thus, all suitable roads were used for spoor-tracking, allowing for five transects of similar length (Figure 3.1).

Previous research has demonstrated that transects should be driven in an east to west direction, early in the morning, as the sun is low and behind the observers, enhancing the shadows and therefore the definition of any tracks (Smallwood & Fitzhugh 1995). Conducting transects over midday, when the sun is directly overhead, reduces the definition of shadows on the ground and therefore the visibility of tracks, and should be avoided (B du Preez, WildCRU PhD candidate, pers. comm.). Thus, in my survey, spoor count transects were started at first light and all transects were sampled with equal frequency and effort (Stander 1998).
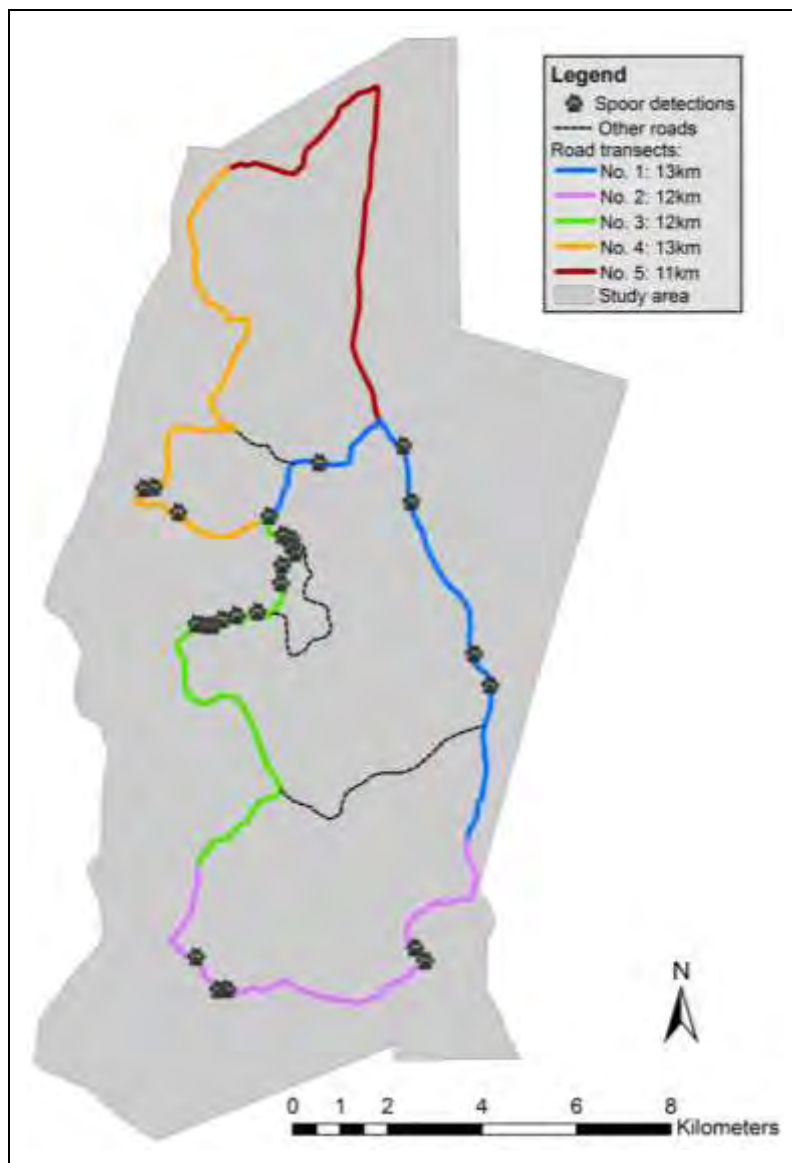
Figure 3.1: The Mangwe study area (200 km²) showing the five road transects used for the spoor counts, and the 27 incidences of leopard spoor that were detected.

An open, short-wheelbase Toyota Landcruiser was used to drive transects at a speed of between 10 and 20 km/hour (Stander 1998, Balme *et al*. 2009a, Houser *et al*. 2009). Three observers (the same two tracking experts and either myself, or another experienced biologist) were seated on the vehicle in a manner aimed at maximising the view of the road. One tracker was seated on the front of the vehicle, the driver on the left (left-hand drive Landcruiser) of the vehicle and the other observer on the right hand side (Figure 3.2).

When fresh leopard spoor was encountered it was measured, photographed alongside a ruler with a sticker indicating the transect and spoor number, and assessed for sex and age class (Figure 3.3). Two sets of measurements were taken for each track; a total width and length measurement at the longest and widest points of the paw print (according to Stuart & Stuart 2000), and the pad length and width based on recommendations by JR Peek (pers. comm.) (Figure 3.3). The pad measurements were considered to be more accurate, due to the splayed toes of the entire track being exaggerated in sandy or muddy soils (Riordan 1998, Henschel & Ray 2003).

Figure 3.2: The open, short-wheel base Toyota Landcruiser used for spoor counts, showing the positions of the three observers.

Figure 3.3: An example of fresh leopard spoor photographed alongside a ruler with a sticker indicating the corresponding transect and spoor number. Red arrows indicate the length and width measurements of the track and the yellow arrows indicate pad length and width measurements, both taken at the widest point of the track.

For each set of fresh tracks encountered from the same individual, five front prints and five hind prints (if available) were measured (du Preez *et al*. 2011). The substrate type at that point was also recorded. Using a GPS, distance from the start of the transect was recorded at each track encountered and a locality fix for each set of tracks, this was later mapped using ArcGIS (Figure 3.1). I tried to assign each set of tracks to a known individual, according to the camera-trapping data and the location of the collared individuals (Balme *et al*. 2009a). However, this was not always possible, as often the nearest cameras did not register any leopard presence, therefore individuals could not always be identified from spoor with absolute certainty. Tracks that were likely to be those of the same leopard were only recorded once per day (Stander 1998, Houser *et al.* 2009). If an individual could not be identified from the track set, any tracks that were found within 500 m and heading in the same direction that were of similar size and of the same sex were assumed to be the same animal and were ignored (Funston *et al.* 2010). To ensure that only fresh tracks were counted each day, a knob thorn (*Vichellia nigrescens)* branch was dragged behind the vehicle each day, creating a clean surface to track the following day (Funston *et al.* 2010). This also served as a way to remove any grass inhibiting visibility and to create a dusty surface for tracking (Smallwood & Fitzhugh 1995).

Spoor density was calculated as the number of individual spoor per 100 km, and spoor frequency was calculated as the number of kilometres per spoor (Stander 1998). Sampling precision was assessed using a bootstrapping analysis (Sokal & Rohlf 1995). This was done by randomly selecting two of the transects and increasing the sample progressively to 3, 4, 5, up to 50, each

time calculating new means and coefficients of variance for spoor frequency (CV) (Stander 1998). The CV was then plotted against effort and the desired sampling intensity was deemed to be at the point where CV reached 20% (Funston *et al.* 2010). Leopard density was calculated by calibrating spoor density with three published calibration equations, which were derived from linear relationships between track density and true carnivore density (Stander 1998, Balme *et al.* 2009a, Funston *et al.* 2010).

3.3.2 Camera-trapping surveys

Equipment

Twenty 'Wildview Xtreme 5' (Wildview, Grand Prairie, Texas) digital cameras were used in this study. These cameras were recommended, based on previous experience of similar studies (C Pearson, Camera Traps Cc, pers. comm.). The camera is triggered by heat- and motion-detecting sensors and boasts other features such as five mega-pixel resolution, 10-meter flash strobe range, a burst mode enabling three sequential photographs to be taken for each 'capture', time and date stamps on each photograph, and timeouts of various durations between bursts. These cameras run off four rechargeable C-Cell batteries, which lasted between four and 12 days. The cameras come with an adjustable strap that can be strapped to trees or mounted to poles fixed in the ground.

Figure 3.4.a: A camera-trap photograph blurred by moisture which obscured the camera lens.

Figure 3.4.b: A camera-trap photograph which has been overexposed by the reflection of the flash off the wooden 'roof'.

No protective covers were used for the cameras in this study because elephants (*Loxodonta africana)*, rhinos (*Diceros bicornis* or *Ceratotherium simum)* and lions (species known to damage cameras) were not present (Karanth & Nichols 1998, Steyn 2006, Chapman & Balme 2010). Although the cameras are waterproof, small wooden 'roofs' had to be used to shelter the lens from moisture in the rainy season, as the moisture obscured the images (Figure 3.4.a). These

roofs had to be placed about 10 cm above the camera otherwise the reflection from the flash would also affect the images (Figure 3.4.b). Cameras were checked every day for the first week of the study in order to check for any leaves or grass that may have inadvertently triggered the camera. Once I was confident with the battery life of the cameras, cameras were checked every four days to replace batteries and to download photographs.
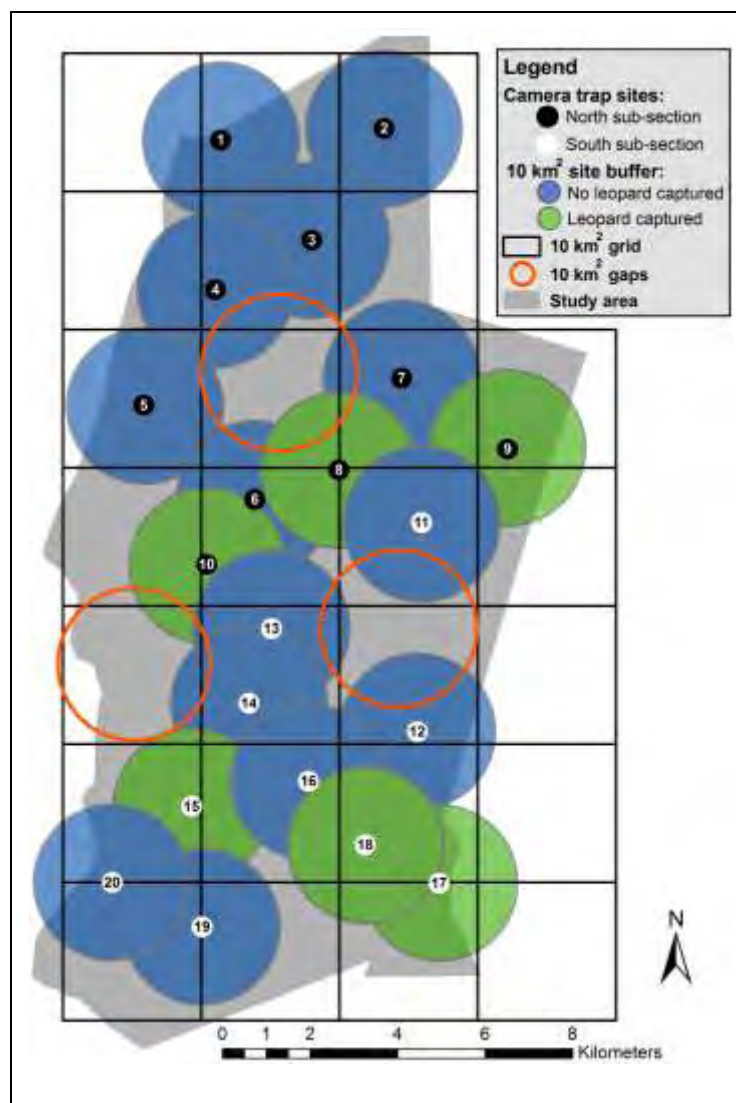
Figure 3.5: The Mangwe study area showing 10 km^2 grid and first (unbaited) camera survey sites split into northern (black points) and southern (white points) sub-sections, orange circles highlighting 10 km^2 gaps.

Unbaited camera-trapping survey

The first camera-trapping survey was conducted between 23 October 2009 and 5 December 2009. This assessment was based on previous studies aimed at individually identifying cryptic carnivores for capture-recapture density estimations (Karanth & Nichols 1998, Henschel & Ray 2003, Silver 2004, Balme *et al*. 2009a). Two cameras were placed at each camera trapping site with the intention of photographing both flanks of the animal as it passed the cameras (Figure 3.6). This results in a 'set' of photographs, one taken by each camera at the same moment that an animal passes the camera trap site. Due to the size of the study area (200 km^2) and the limited number of cameras (20), the area was split into two contiguous subsections which were sampled sequentially (Figure 3.5) (Karanth & Nichols 2002, Silver *et al.* 2004, Balme *et al*. 2009a). Each section was sampled for a total of 20 days, with one sampling occasion defined as a period of 24 hours starting at 15h00. Although the two sub-sections were sampled sequentially, the total number of 'captures' for Occasion One (the first 24 hour period, or day one) would include the total number of 'captures' and 'recaptures' from the first occasion of *each* sub-section sampled (Karanth & Nichols 2002, Silver *et al*. 2004). Therefore, the total number of captures for Occasion Two would be the sum of captures and recaptures from the second day of both sub-sections, and so on (Karanth &

Nichols 2002, Balme *et al*. 2009a). This resulted in a total of 20 sampling occasions for the entire 40-day camera-trapping survey.

Site selection

The cameras were placed at sites intended to maximise the chance of leopard captures, whilst satisfying the assumption that no animal had zero probability of being photographed (Karanth & Nichols 1998, Silver 2004, Balme *et al*. 2009a). To ensure that there were no gaps within which a leopard could reside without being detected, at least one trap site was placed within the minimum recorded home range size area for a leopard. In this case, the minimum recorded home range size for leopards located in similar terrain is 10 km^2 (Smith 1977). A 10-km^2 grid was drawn over the hard-copy topographic map for the area (recreated here for illustrative purposes; Figure 3.5) and camera sites were chosen within each 10-km^2 grid square. Sites within the squares were chosen according to areas believed to be frequented by leopards, based on a previous spoor-tracking pilot study, local information and other similar studies (Karanth & Nichols 2002, Silver *et al.* 2004, Steyn 2007, Balme *et al*. 2009a).

Camera placement

Once these sites were chosen, cameras were set up perpendicularly along existing game trails, roads or river beds. Cameras were set approximately one to two meters off the path where the leopard was expected to pass (Silver *et al.* 2004) (Figure 3.6). This allowed the camera to focus if the photograph was taken when the leopard was straight in front of the camera. Once a suitable tree near the path was found, a camera was strapped to it approximately 60 cm (the average shoulder height of a leopard) off the ground (Silver 2004). To allow for the slow trigger speed, cameras were aimed slightly down the path to allow the shutter speed as much time as possible to trigger, in order to photograph the passing animal (Figure 3.6). This slight offset also prevented mutual flash interference from the opposite camera (Karanth & Nichols 1998, Silver *et al.* 2004).
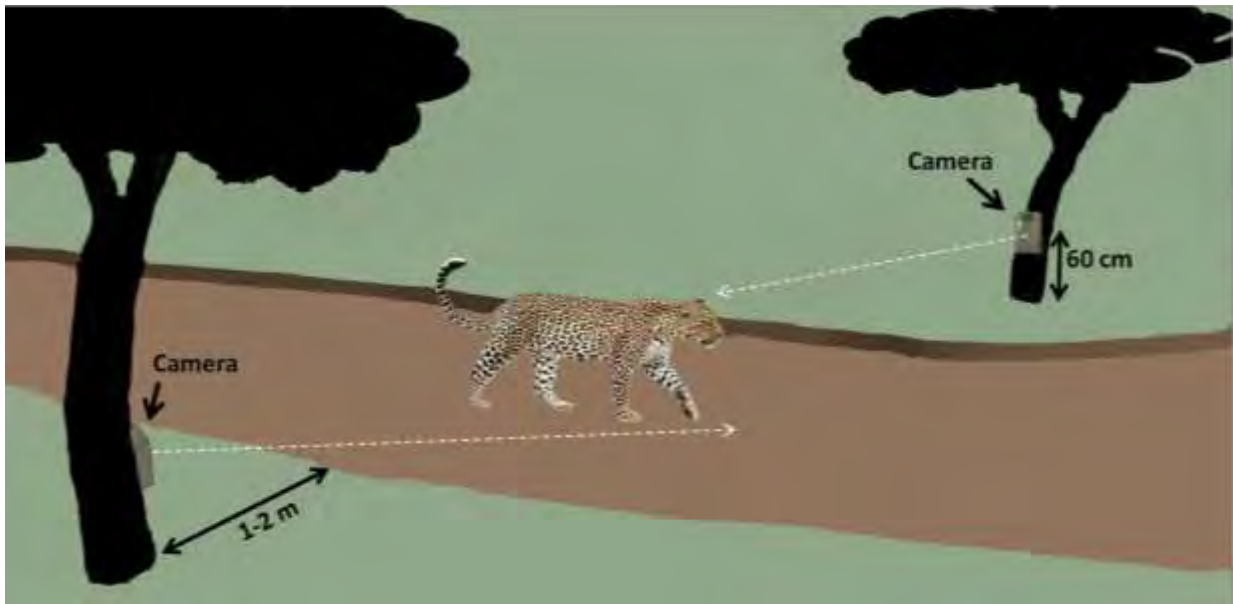
Figure 3.6: Camera placement used in the unbaited survey. Cameras were set 1-2 m off the path and approximately 60 cm (average shoulder height of leopard) off the ground, facing each other, although slightly offset to prevent mutual flash interference. Cameras were aimed slightly down the path (dashed white arrows).

Due to the slow trigger speed (5 seconds) of the cameras, this method was not successful in capturing identifiable leopard photographs (Figure 3.7). However, it did provide a useful inventory of the mammals of the area, for which no published data exist. These data will not be presented here, but are attached as an appendix (Appendix II) and will be published as a separate journal article. The leopard photographs from this survey were therefore not used in any further analyses.

Figure 3.7: Examples of unidentifiable photographs obtained during the unbaited camera survey.
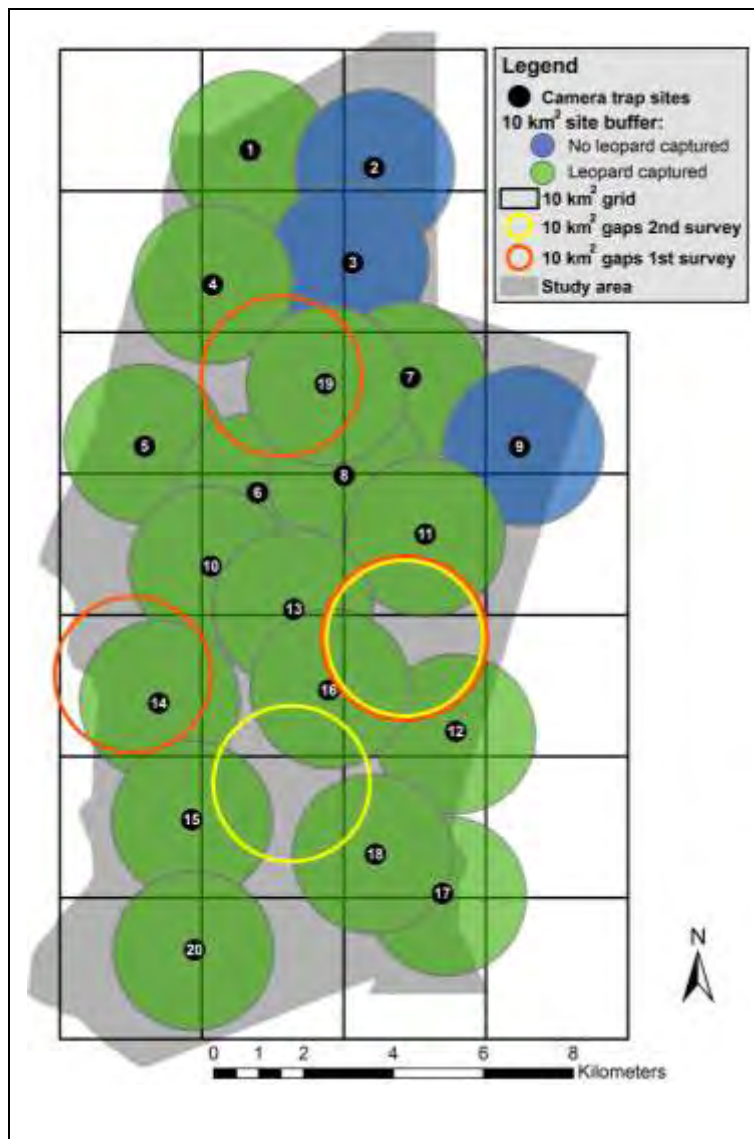
Figure 3.8: The Mangwe study area showing the 10-km² grid and the second camera survey sites, orange circles highlighting 10-km² gaps from the first survey, and yellow circles highlighting gaps in the second survey.

Baited camera-trapping survey

Based on the successful results of an ongoing study in the Save Valley Conservancy in south-east Zimbabwe (Joubert 2009), and because of the poor results of my initial unbaited survey, I decided to employ a baited camera-trapping technique. The second camera-trapping survey was conducted between 26 June 2010 and 1 September 2010 (66 days). This survey employed a slightly different method in that the baited camera sites only required one camera per site. This enabled all 20 sites to be deployed simultaneously.

Site selection

Most of the sites used were the same as the unbaited survey; however, a few sites were moved to more productive areas, and to try and reduce any gaps that existed in the first camera survey layout (Figure

3.8). Ten of the original 20 sites remained the same as the unbaited survey; five were relocated but within the same 10-km² block from 400 m up to 1.1 km away from the initial site; four were relocated into the adjacent block, from 800 m up to 2.4 km away from their initial position, and one camera was moved to a new site 12.5 km away (Site 19). Figure 3.8 shows these new sites as well as the 10-km² gaps in the first survey (orange) and the gaps that resulted in the second survey (yellow). Although there were still gaps in which, theoretically, a leopard could exist without being detected, these areas contained villages which are likely to deter leopards from residing in these areas.

Camera placement

Once the sites were chosen, a suitable 'bait tree' was located. This tree required a fork low enough to rest the 'leader pole' onto, so that the leader pole was at an incline gentle enough for the leopard to climb easily. The leader poles were at least 20 cm in diameter and placed at angles of 45° or less, against the bait tree (Figure 3.9). Cattle foetuses, donated by local abattoirs, were used as bait. This was advantageous as no additional wild animals or livestock had to be killed for bait, and there was always a constant supply of foetuses from the abattoirs. This method is more successful, as leopards are detained at the bait, in front of the camera, long enough to obtain a clear identification photograph (Joubert 2009). Sites were checked every day or every second day to replenish baits if needed, and to check the camera batteries and download photographs.
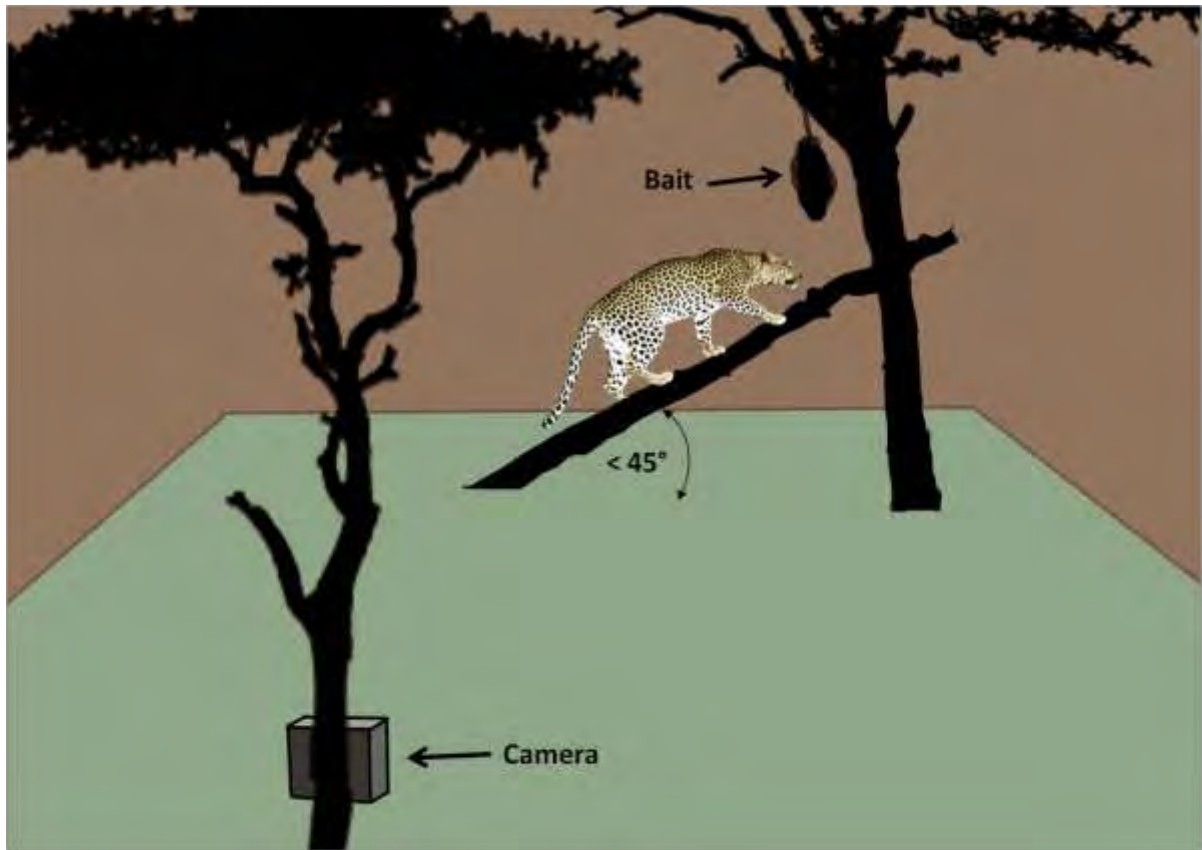
Figure 3.9: Camera placement used for the baited survey. A pole was placed against the bait tree at an angle suitable for the leopard to easily climb (< 45°). A single camera was attached to a tree 2-3 m away, aimed just below the bait.

Data analysis

In order to analyse the data, individual leopards had to be identified in all the photographs, based on their individual pelage patterns (Miththapala *et al.* 1989, Karanth & Nichols 1998, Henschel & Ray 2003). Following guidelines by Heilbrun *et al.* (2003) and Jackson *et al.* (2006) a photograph was considered an initial capture if it could not be positively matched to any previous photographs; if the leopard was unidentifiable due to a poor photograph it was not counted as a capture. Individual markings were identified and assigned as a primary feature for each individual (Figure 3.10). These were usually the most clearly visible and distinct markings, all other markings were classified as secondary features.

a.
b.
c.
d.
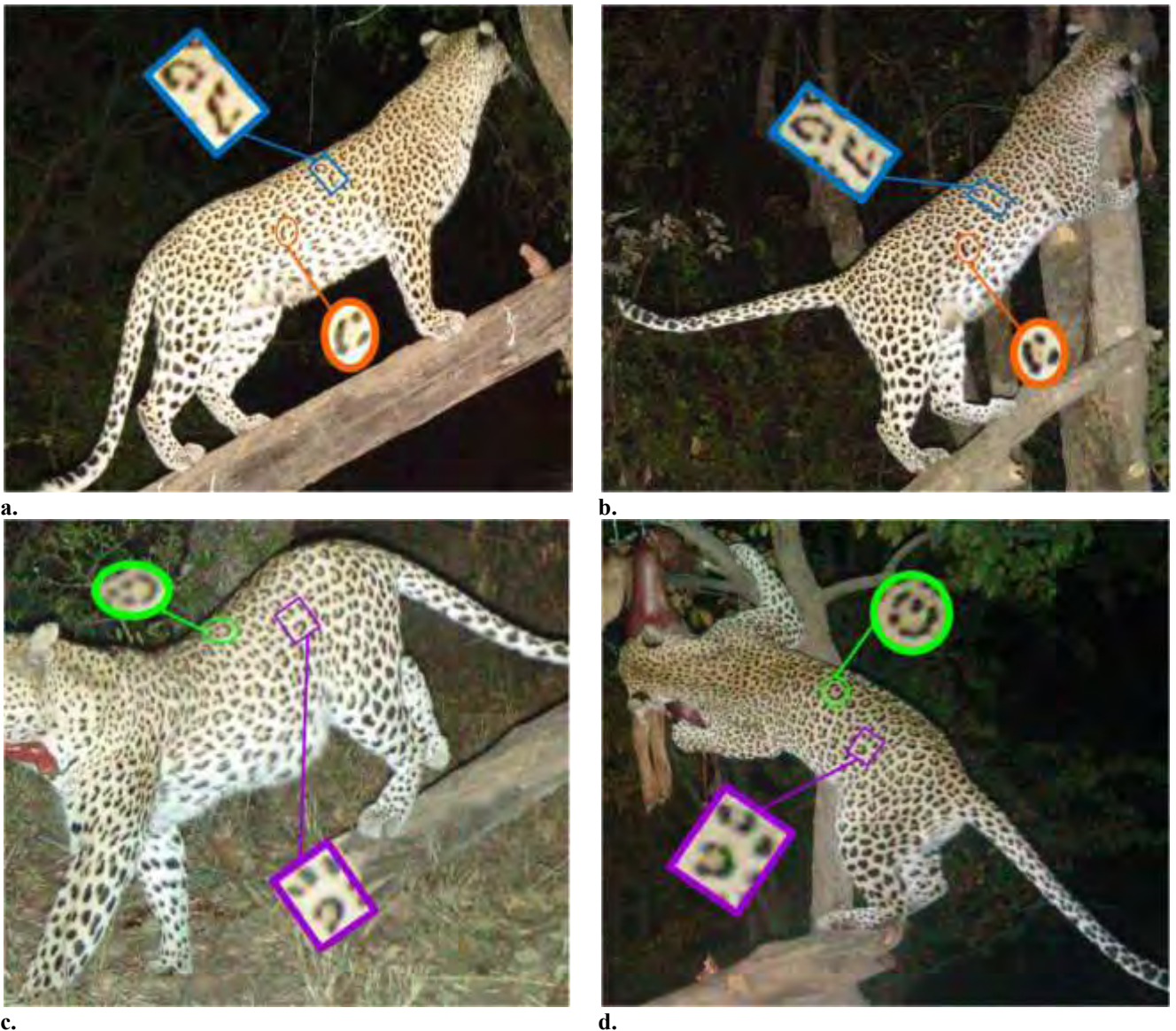
Figure 3.10.a and b: Two right-side photographs captured of the same individual at different baited-camera sites illustrating the primary identifying feature (blue rectangle) and a secondary feature (orange circle). **Figures 3.10.c and d** illustrate the same individual's left side, with the green circle outlining the primary left-side feature and the purple rectangle illustrating a secondary identifying feature.

A positive identification was made by comparing the primary feature and at least one secondary feature. Identification of one different feature was considered sufficient to determine that two photographs represented two different individuals (Figure 3.11). Individual leopards were sexed and aged based on any sign of scrotal testes (for males), and their size judged from the photograph, as well as from any tracks found at the site, with males' tracks being considerably larger than females' (Stuart & Stuart 2000).

CAPTURE:

A capture history was then created for all sampling occasions to analyse the capture data. In this survey, one sampling occasion was defined as a 24-hour period, starting at 15h00 each day. A binary matrix was then compiled (Appendix III), displaying a 1 for a positive capture and a 0 for no capture, for all sampling occasions. These data were analysed in the programme CAPTURE (Otis *et al.* 1978, Rexstad & Burnham 1992), following the procedures described by Otis *et al.* (1978) and White *et al.* (1982), using models developed for closed population capture-recapture abundance estimates (Karanth & Nichols 1998, Wang &

Figure 3.11: The top photograph depicts the female identified in Figure 3.10, the left inset highlighting her primary identification feature. The bottom photograph depicts a different female. By comparing the same area of her pelage, one can immediately determine that this is not the same individual.

Macdonald 2009). CAPTURE calculates estimates of population abundance under seven probabilistic models that differ in their assumed sources of variation in capture probability (Rexstad & Burnham 1992). These differences in capture probability include the behavioural response (M_b) of leopards to camera-trapping (e.g. trap avoidance or trap encouragement, through baiting), time specific (M_t) variation (e.g. influences through changes in weather conditions), and heterogeneity (M_h) among individual leopards (e.g. a larger home range could

expose that individual to more trap sites) (Otis *et al.* 1978, Trolle & Kery 2003, Wang & Macdonald 2009). The simplest model, M_0 (the Null Model), assumes a constant capture probability over all occasions and animals (Otis *et al.* 1978, White *et al.* 1982, Trolle & Kery 2003).

CAPTURE computed goodness-of-fit, and between-model test statistics for each model, providing information about the appropriateness of each model. The programme includes a model selection algorithm that uses a discriminate function to provide an objective criterion for selecting the most suitable model (Rexstad & Burnham 1992). CAPTURE also computed a closure test statistic from the x matrix to test for population closure.

Buffer strip methods:

The resulting abundance estimates from CAPTURE were used to calculate leopard population density in the study area. Density was defined as *D= N/A(W)*, where *N* is the leopard abundance and *A* is the area where the animals were sampled, including a buffer width (*W)* around this area (Karanth & Nichols 1998). Numerous studies have assessed the most accurate method of calculating this effectively sampled area (Wilson & Anderson 1985, Karanth & Nichols 1998, Silver 2004, Soisalo & Cavalcanti 2006, Balme 2009a). As the area demarked by the trapping grid of the camera layout - or the outer perimeter (or polygon) - is not necessarily the entire area in which the sampled animals range (Otis *et al*. 1978, White *et al*. 1982), a boundary strip is usually added to the polygon and is defined by the outermost trap sites (Karanth & Nichols 1998). I used Balme *et al.*'s (2009a) method for calculating the boundary strip in this study, based on the analysis of five different methods for calculating the buffer strip, they found that the mean maximum distance moved outside of the study area (MMDMOSA) by radio-collared leopards was the most accurate method of estimating an effectively sampled area. The traditionally used half the mean maximum distance moved (HMMDM) by individual leopards, to more than one camera-trapping station (Wilson & Anderson 1985, Karanth & Nichols 1998, Trolle & Kery 2003, Soisalo & Cavalcanti 2006) resulted in the second best performing method (Balme *et al*. 2009a). Balme *et al*. (2009a) also found that by creating a circular band around each individual camera station (based on Silver *et al.* (2004), where the radius is equal to HMMDM or MMDMOSA, resulted in a more accurate estimate of the effectively sampled area, compared to adding the buffer strip to the minimum convex polygon (MCP) created by the outer camera sites.

ArcGIS was used to calculate all the parameters required for the boundary strips and the resulting effectively sampled area. The MMDMOSA method involved measuring the maximum

distance moved by the collared leopards, outside of the area delineated by the outer traps (Figure 3.15) (Balme *et al.* 2009a). GPS telemetry data from the three collared leopards (one male and two females) in the current study were used to estimate these distances (see Chapter 4). The resulting distance served as the radius length to create buffers (which serve as a proxy for home ranges) around each camera-trap site in the area (Silver *et al.* 2004, Balme *et al.* 2009a). Once these buffers were merged, the effectively sampled area was calculated. The other method involved measuring the maximum distance moved (MMDM) by individuals photographed on more than one occasion, at more than one trap site, during the camera survey, and halving this distance (HMMDM) (Figure 3.15).

SPACECAP:

Due to the *ad hoc* approach for estimating the effectively sampled area, and the lack of a theoretical justification for the method, a new, more statistically robust programme (SPACECAP), developed specifically for the analysis of population density estimates using Spatially-Explicit Capture-Recapture (SECR) models, was also used (Singh *et al.* 2010). SPACECAP directly estimates animal density by explicitly using the data from the capture histories in combination with the co-ordinates of the captures (trap sites) under a Bayesian modelling framework (Singh *et al.* 2010). SECR models are a recent advance in the field of population density estimation and are based on the initial work of Borchers & Efford (2008), Royle & Young (2008) and Royle *et al.* (2009) and Sollmann *et al.* (2011). Using the spatial data of the capture histories, the model first determines an individual's activity centre, and then estimates the density of these activity centres across a precisely defined area containing the trap array (Royle *et al.* 2009, Sollmann *et al.* 2011). By incorporating the spatial locations of the captures and the array of the traps supposedly circumvents, the problem of estimating the effectively sampled area (Singh *et al.* 2010). Another advantage of this model is that, through the Bayesian framework, it offers non-asymptotic inferences which are more appropriate for small samples of capture data, which are typical of most camera-trapping studies (Singh *et al.* 2010).

SPACECAP runs as a package through R (The R Foundation for Statistical Computing 2011). Three input files are required to run the analysis, and the data has to be entered precisely, according to strict formatting guidelines. The first file is the 'animal capture details' (Appendix IV) – this file contains fields for each leopard's identification number, the trap number where it was captured and the sampling occasion number of the capture incidence. My data were sorted by the sampling occasion column. The second file is the 'trap deployment details' (Appendix V) – this file is essentially a matrix represented in binary format (0/1) that indicates which particular camera stations were operational. To construct this, columns are needed for the trap site

numbers, and their corresponding x and y UTM co-ordinates. All sampling occasions are then entered as column headers, so that each cell below will show a corresponding relationship between the trap site (row) and sampling occasion (column). This relationship was expressed either as a 1 if a camera station was in operation on a specific sampling occasion, or a 0, if for some reason, the camera was off (e.g. due to flat batteries, damage, theft). The last input file is the 'potential home range centres' file, which requires spatial data analysed in GIS software. A large number of equally spaced points, in the form of a very fine mesh, are needed to represent all possible potential activity centres (or home range centres) of the individuals – the resulting area incorporating this mesh is known as the 'state-space' (Singh *et al.* 2010).

In ArcGIS, I created a rectangle the 'minimum area rectangle', by connecting all the outermost camera-trap sites (Figure 3.12). This minimum area rectangle was then buffered to create the 'state-space' area. The buffer distance had to be large enough to ensure that individuals outside the buffered area had a zero probability of being photographed by any of the camera-traps within the study area (Singh *et al.* 2010). To calculate the buffer area, I averaged all the GPS telemetry co-ordinates used to calculate the collared male's home range in order to find the centre point of this range. I overlaid this average co-ordinate over the rest of his GPS points in ArcGIS. I then measured the furthest collar fix point from this average co-ordinate, as this would be the maximum distance moved (MDM) from anywhere in the male's home range during the study. If more than one male had been collared I would have used the average maximum distance moved of all males to calculate the buffer distance, as the sample size of collared individuals could greatly influence this measurement (Balme *et al*. 2009a). However, the buffer distance used here is likely to be large enough to exclude any other leopards whose home ranges fell outside of the buffer area from being captured in the study. This is based on comparisons with similar telemetry studies (Mizutani & Jewell 1998, Marker & Dickman 2005a, Swanepoel 2008, Steyn 2007), where the average male MCP home range was similar (see Chapter 4). The entire area encompassed by the buffered area was therefore considered the 'state-space', and numerous equally spaced points representing 'home range centres' were then generated for this extended area (for a detailed description of how these data are created, see Appendix VI).
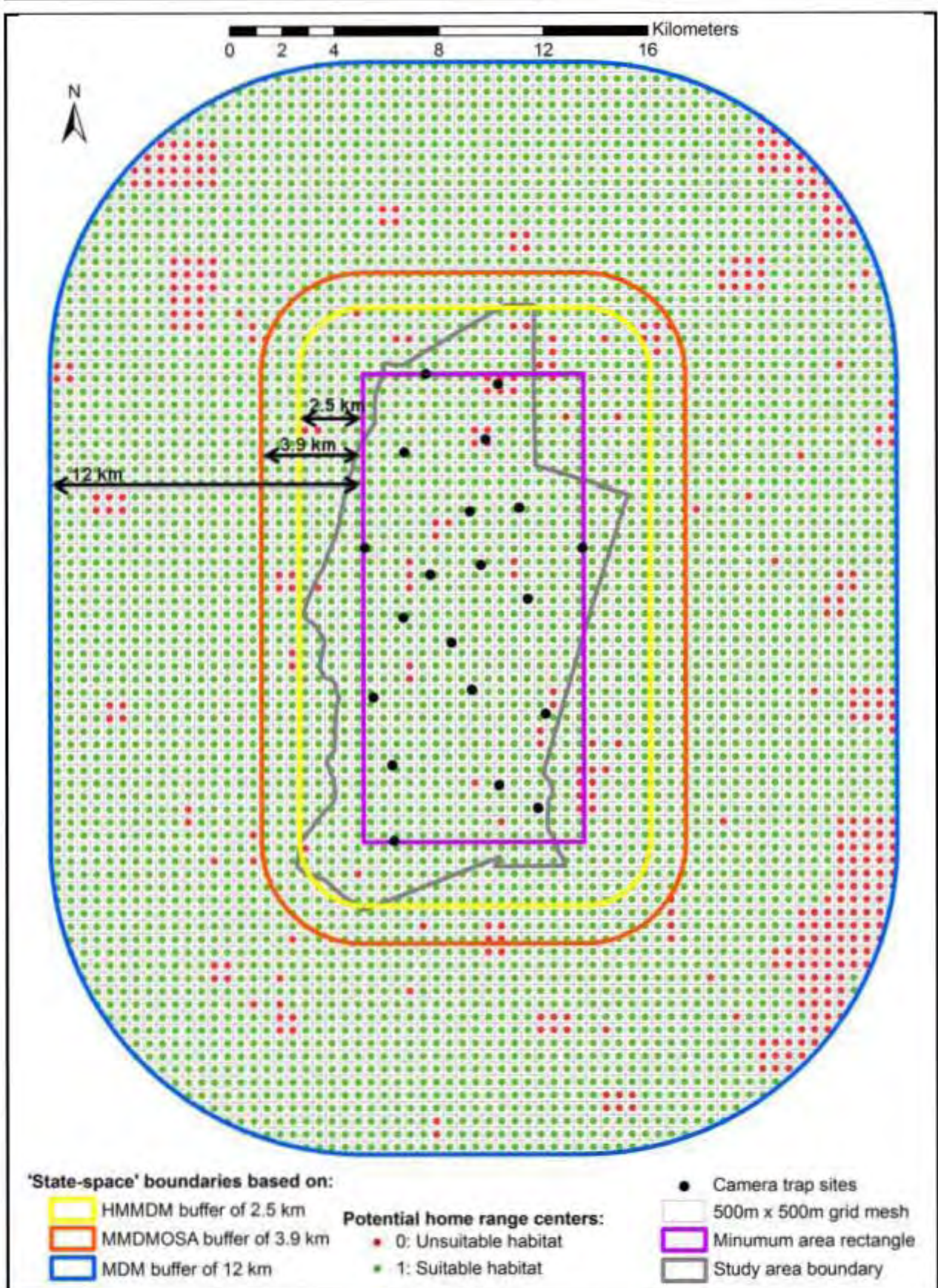
Figure 3.12: An example of the spatial data created in ArcGIS for the third input file 'potential home range centres' for the programme SPACECAP. The red points represent unsuitable leopard habitat, whilst the green represent suitable habitat, as determined by a topographic map and Google Earth satellite images. The 'state-space' boundaries were calculated using various buffer distances of 2.5 km (HMMDM), 3.9km (MMDMOSA) and 12 km (MDM).

For this analysis, I created a mesh of equally spaced points that were 500 m apart; this distance was based on information from Sollmann *et al.* (2011) and Royle *et al* (2009). In reality, many of these 'home range centres' will fall over areas that are not suitable habitats for leopards (e.g. bodies of water such as dams or lakes), and these areas were identified by overlaying the mesh of equally-spaced points over topographic maps or aerial photos for the area. Once any unsuitable habitats were identified and marked, the data were entered into the final 'potential home range centres' file with columns for the x and y co-ordinates (UTM) and a column titled 'Habitat' (Appendix VII). The 'habitat' column was the resulting data from identifying suitable home range centres in ArcGIS, whereby a 1 represented suitable habitat, and a 0 represented unsuitable habitat. Analyses were also run using the same buffer methods as those used for CAPTURE (MMDMOSA and HMMDM), providing density estimate comparisons from three different 'state-space' areas. The SPACECAP input files were then uploaded into the programme and the appropriate model combinations were chosen for analysis (Singh *et al.* 2010). I used the following model definitions: Trap Response Present, and Spatial Capture-Recapture. For the Markov-Chain Monte Carlo (MCMC) parameters, I used: 50 000 for the Number of Iterations, 1000 for Burn-in, no Thinning rate was selected (therefore a value of 1) and Data Augmentation of 26 (Singh *et al.* 2010, Sollmann *et al.* 2011).

3.4 Results

3.4.1 Spoor index survey

Five transects, totalling 61 km, were sampled 10 times, which resulted in a total distance surveyed of 610 km and a penetration rate of 3.3 (Table 3.2). The total number of spoor detected was 27, six of which were male, and the remaining 21, female. This resulted in a spoor frequency (km/spoor) of 22.59 and a spoor density (spoor/100 km) of 4.43.

Table 3.2 Summary of the statistics from the spoor index survey in the Mangwe area.

Survey area	200 km²
Total transect length (5 transects)	61 km
Total distance driven	610 km
Penetration rate	3.3
Total leopard tracks recorded	27
Spoor frequency (Km/spoor)	22.59
Spoor density (Spoor/100km)	4.43
Male leopard tracks	6
Female leopard tracks	21

Sampling precision, as measured by the coefficient of variation (CV), increased rapidly as the CV decreased with an increase in sample size. CV reached 20% at 24.7 spoor counts (Figure 3.14.a). Variability decreased by a further 2.5% at the total number of tracks counted (27), reaching a CV of 17.5% at the end of the survey. The rapid decrease in variability observed in the first 220 km driven (Figure 3.14.b) mirrors that of the first 10 spoor counts (Figure 3.14.a). The distance driven at the point where 24.7 spoor were counted, is 530 km with a CV of 7.5% (Figure 3.14.b). An asymptote was not reached for spoor counts, suggesting that a greater sampling effort was needed.
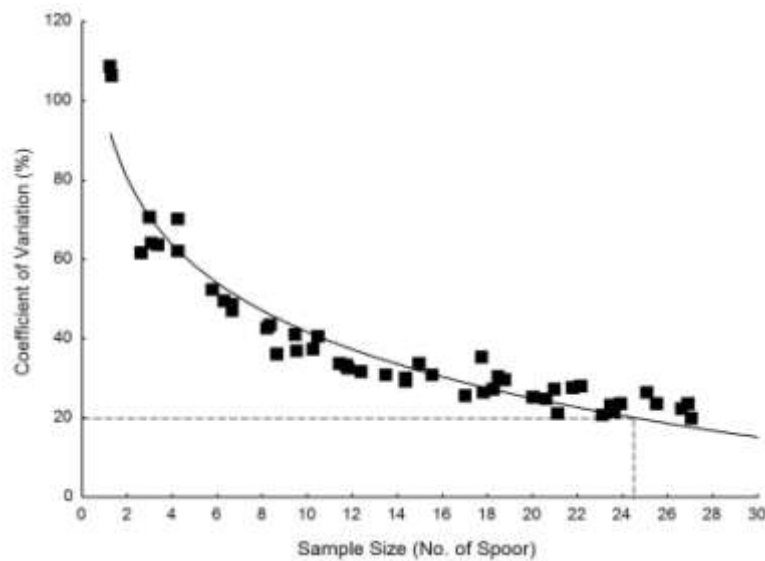
Figure 3.14.a: The relationship between sampling precision as measured by the percentage coefficient of variation and spoor counts (SD as the percentage of the mean). CV of 20% was reached at 24.7 spoor samples.
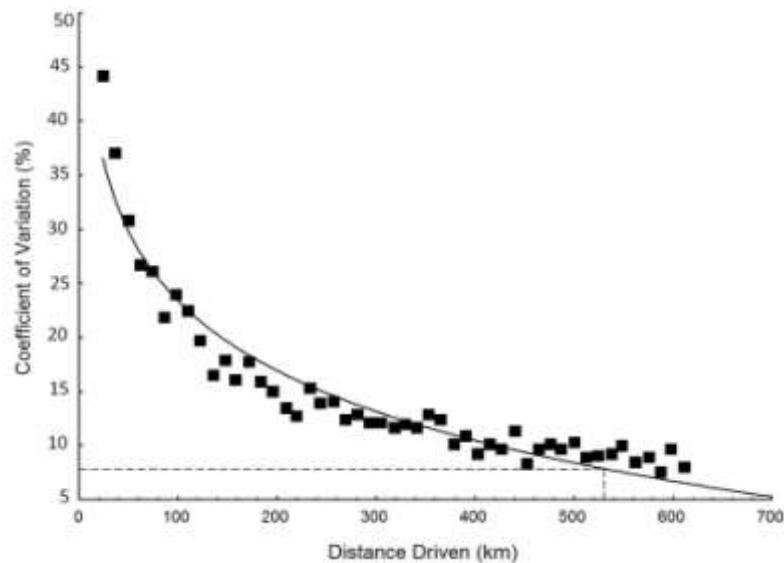
Figure 3.14.b: The relationship between sampling precision as measured by the percentage coefficient of variation and distance driven during spoor counts (SD as the percentage of the mean). When spoor count reached 24.7 (Figure 3.14.a), the distance driven at this point was 530 km, resulting in a CV of 7.5%.

Estimated population density of leopard varied according to the calibration equation used (Table 3.3). The spoor density of 4.43 tracks/100 km, derived from the transects, was used to solve these equations. The highest density of 3.29 leopards/100 km^2 resulted from Balme *et al.*'s (2009a) Phinda equation. The next highest density of 2.33 leopards/100km^2 resulted from Stander's (1998) equation which was calibrated for leopards in the Tsumkwe District of Namibia. The lowest density of 1.28 leopards/100 km^2 resulted from Funston *et al.*'s (2010) combined model which was calibrated for a combination of lion, cheetah, leopard and hyaena data from studies conducted on sandy soils.

Table 3.3: Estimated population densities of leopards (no. of leopards/100 km^2) derived from the spoor density results using three calibration equations.

Method	Equation	Density (leopards/100 km^2)
Stander (1998) leopard equation	$y = 1.9x$	2.33
Balme *et al.* (2009a) Phinda calibration	$y = 1.33x + 0.05$	3.29
Funston *et al.* (2010) Combined model (all carnivores) on sandy soils	$y = 3.15x + 0.40$	1.28

3.4.2 Camera-trapping surveys

Unbaited camera-trapping survey

A total effort of 800 trap nights (20 cameras x 40 survey nights) was achieved, producing 507 photographic events of all mammals, 54% (*n* = 269) were of cattle, humans (*Homo sapiens sapiens*), donkeys (*Equus asinus*), and domestic dogs (*Canis familiaris*), and are excluded from further results (Table 3.4). From the remaining photographs 30 mammal, 11 bird and one reptile species were identified (Appendix II). The unbaited camera-trapping methodology did not prove to be successful for capturing identifiable photographs of leopards and resulted in only six leopard photographs. The trigger speed of the cameras used, appeared to be too slow for the opportunistic capture of passing leopards, as leopard tracks would often be observed near the camera-traps, but no photographs were taken. Of the six leopard photographs, only two sets could be identified as the same individual, the rest could not be distinguished from one another. However, when these leopard photographs were compared to the more successful photographs obtained during the baited camera survey seven months later, the four remaining photographs were successfully matched to individuals identified in that survey.

Baited camera-trapping survey

The sampling effort achieved for this survey was 1320 trap nights (20 cameras x 66 survey nights), producing a total of 1713 photographic events of mammals, with only 15% (n = 248) of anthropogenic influence (Table 3.4). The baited survey produced 292 leopard photographs, compiling 13 individual leopards (eight females and five males) captured on 95 separate occasions. One set of leopard photographs were excluded from analysis as the leopard could not be identified.

Table 3.4: Summary of the unbaited and baited camera-trapping surveys conducted in the Mangwe area.

	Unbaited		Baited	
	n=	*%*	*n=*	*%*
No. of camera-trapping days	800	100	1320	100
Total no. of photo captures	507	100	1713	100
Cattle, humans, donkeys, domestic dogs	269	54	248	15
Total leopard photo events	6	1	292	22
No. of individual leopards identified	1		13	
Total carnivore species	12		10	
Total carnivore photo events	72	14	1260	74
Total mammal species	30		23	
Total medium-large mammals out of possible 37 'known' species	26	70	21	57
Total bird species	11		13	
Total bird photo events	23	5	106	6
Total reptiles	1		0	

CAPTURE

Capture frequencies were calculated from the individual capture histories of each leopard, which were entered into the capture matrix used for the CAPTURE software analyses (Appendix III). Capture frequencies ranged from 2 to 27 per individual, with an average of 7.31 ± 8.06. The number of captures per sampling occasion ranged from one to five, with an average of 2.11 ± 1.27 per sampling occasion.

The best-fitting model selected by CAPTURE was the M_h model (criterion score of 1.0) which assumed that each individual had its own capture probability and that this remained constant over the sampling period (Karanth & Nichols 1998). The estimated leopard population size for this model using the jackknife estimator was 15 ± 2.60, with a capture probability per sampling occasion of 0.064. The second best performing model selected by CAPTURE was the M_o model (criterion score of 0.96), which assumed constant capture probability over all occasions and animals (White *et al*. 1982, Trolle & Kery 2003). The M_o model resulted in a population size of 12 ± 0.31 and an estimated capture probability of 0.081. However, population closure was not confirmed by CAPTURE (z = -3.0588, p = 0.0011).
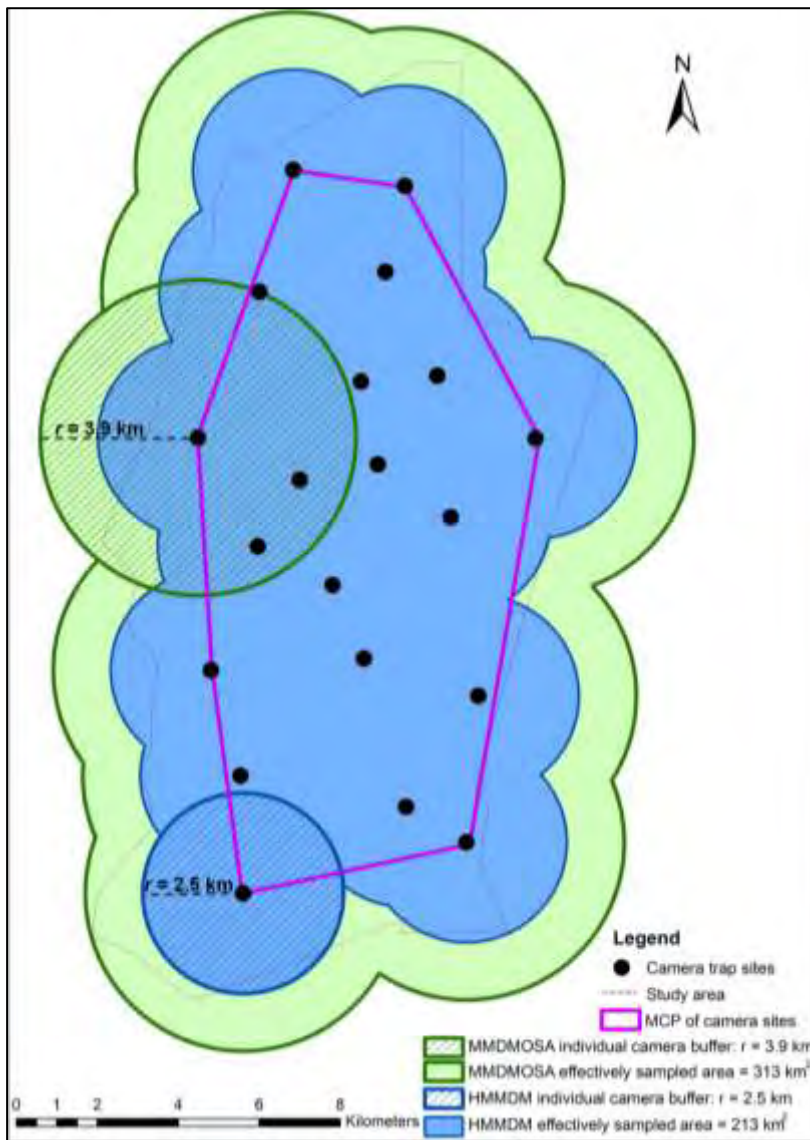
Figure 3.15: The effectively sampled areas of 313 km² and 213 km² respectively, calculated using individual camera-trap buffers and the MMDMOSA (3.9 km) and HMMDM (2.5 km) methods.

Buffers

The MMDMOSA was 3.9 km, which was the radius used to buffer each camera-trap site to create a merged polygon representing an effectively sampled area of 313 km² (Figure 3.15). This buffer method resulted in a leopard density estimate of 4.79/100 km² for the M_h model and 3.83/100 km² for the M_o model (Table 3.5). The HMMDM of nine leopards was 2.5 km, and when used as a radius to buffer each camera site (Figure 3.15) this produced an effectively sampled area of 213 km² (68% of the MMDMOSA area). The resulting leopard density estimates were 7.04/ 100 km² for the M_h model and 5.63/ 100 km² for the Mo model (Table 3.5).

Table 3.5: Estimated leopard densities (no. of leopards/100 km^2) calculated in the programmes CAPTURE and SPACECAP using various buffer methods in the Mangwe area.

Programme	Model	Buffer	Effectively sampled area/ State area	Density (Leopards/ 100 km^2)	95% CI
CAPTURE	M_h	3.9 km (MMDMOSA)	313 km^2	4.79±0.83	4.15-7.67
CAPTURE	M_o	3.9 km (MMDMOSA)	313 km^2	3.83±0.1	-*
CAPTURE	M_h	2.5 km (HMMDM)	213 km^2	7.04±1.22	6.10-11.27
CAPTURE	M_o	2.5 km (HMMDM)	213 km^2	5.63±0.15	-*
SPACECAP	Trap response present Spatial Capture-Recapture	2.5 km	303 km^2	5.12±0.62	4.39-6.41
SPACECAP	Trap response present Spatial Capture-Recapture	3.9 km	405 km^2	2.12±0.34	1.65-2.8
SPACECAP	Trap response present Spatial Capture-Recapture	12 km	1237 km^2	1.37±0.23	0.96-1.7

*CAPTURE does not calculate 95% CI for the M_o model

SPACECAP

Three SPACECAP analyses were run using different buffer sizes in order to create the 'state-space' boundaries, the results of which are presented in Table 3.5 (with the CAPTURE estimates) and Table 3.6 (with other parameters calculated by the programme). The density results of the SECR SPACECAP model varied according to the size of the buffer used to create the 'state-space'. However, the initial encounter probabilities (p1) and recapture probabilities (p2) remained constant for all analyses (Table 3.6). A density of 5.12 ± 0.6 leopards/100 km^2 was estimated using a buffer of 2.5 km, a 3.9-km buffer provided a density of 3.9 ± 0.3 leopards/100 km^2, the largest buffer of 12 km resulted in a density estimate of 1.37 ± 0.2 leopards/100 km^2 (Table 3.5). Total number of individuals (Nsuper) estimated by SPACECAP also varied significantly according the state-space used, these estimates ranged from 15 to 32 leopards (Table 3.6).

Table 3.6: The results from the SPACECAP analyses using three different buffer distances to create the state-space.

Buffer used	Variables*	Mean	SD	95% Lower HPD Level	95% Upper HPD Level
2.5-km buffer = state space area of 303 km²	sigma	0.7155	0.1802	0.4286	1.072
	lam0	0.0161	0.0051	0.0072	0.0271
	beta	2.0751	0.2976	1.5042	2.6559
	psi	0.3944	0.0877	0.2288	0.5677
	Nsuper	15.1816	1.8479	13	19
	Density	5.1203	0.6232	4.3845	6.4081
	p1	0.016	0.005	0.0072	0.0268
	p2	0.8689	0.0386	0.7883	0.9361
3.9-km buffer = state space area of 405 km²	sigma	0.8096	0.1967	0.4613	1.2007
	lam0	0.0153	0.0047	0.0074	0.0253
	beta	2.1502	0.2789	1.5895	2.7015
	psi	0.4307	0.0998	0.2413	0.6266
	Nsuper	16.6718	2.6374	13	22
	Density	2.1197	0.3353	1.6529	2.7972
	p1	0.0152	0.0046	0.0073	0.025
	p2	0.879	0.0341	0.8079	0.9387
12-km buffer = state space area of 1237 km²	sigma	0.9239	0.3018	0.4769	1.5227
	lam0	0.0198	0.0056	0.0106	0.0309
	beta	1.9809	0.275	1.4732	2.5297
	psi	0.7936	0.1407	0.5297	1
	Nsuper	31.5184	5.2447	22	39
	Density	1.3722	0.2283	0.9578	1.6979
	p1	0.0196	0.0055	0.0105	0.0304
	p2	0.8568	0.0398	0.7777	0.9252

*sigma – range parameter of an individual; **lam0** – expected encounter frequency; **beta** – the regression coefficient of behavioural response; **psi** – ratio of number of individuals present within the state-space to the maximum number stated in the model; **NSuper** – total number of individuals; **Density** – number of individuals/100 km²; **p1** – encounter probability of an individual before initial encounter; **p2** – encounter probability of individual after initial encounter.

3.5 Discussion

Capture-recapture models are based on three key assumptions that need to be met. The high probability of violating these assumptions in a free-ranging biological population has resulted in various methodologies, all with their own merits and drawbacks, and producing varying density estimates (Karanth *et al.* 2010). This highlights the need for a standardised, comparable method of density estimation (Balme *et al.* 2009a, Sollmann *et al.* 2011). In my study, the spoor count index density estimates varied between 1 and 3 leopards/100 km^2, whereas the camera-trapping density estimates ranged between 1 and 7 leopards/ 100 km^2.

3.5.1 Spoor indices

Effort

Sampling design is crucial for conducting successful field-based carnivore density assessments (Smallwood & Fitzhugh 1995, Stander 1998). Sampling intensity is a key step in this planning; the effort spent must adequately meet the precision requirements for a robust statistical analysis, whilst still being efficient in time and expenditure (Stander 1998, Davidson 2009). Effort required in this study was initially based on calculating an adequate penetration rate; a penetration rate resulting in a number < 7 seemed appropriate, based on previous studies (Stander 1998; Davidson 2009, du Preez *et al.* 2011). The penetration rate achieved in the current study compares favourably with those achieved in other studies e.g. 1.8 − 6.5 (Stander 1998), 3.34 (Houser *et al.* 2009), 3.8 − 7.3 (Davidson 2009) and 2.77 − 7.56 (du Preez *et al.* 2011).

Once road length (as determined by the penetration rate) is determined, the sampling effort along transects can further influence the accuracy and precision of spoor frequency estimates (Stander 1998). Therefore, sampling precision was also quantified by plotting the coefficient of variation of spoor frequency against effort (the number of spoor counts and the total distance driven) (Stander 1998, Funston *et al.* 2010). The desired CV of 20%, as recommended by Funston *et al.* (2010), was achieved at 24.7 spoor counts, although an asymptote was not reached by the end of the survey, suggesting that more effort was needed to achieve higher levels of precision. At the desired level of CV for spoor counts, the distance driven was 535 km, although the level of precision at this point is high, an asymptote was not reached, reflecting the same results as the spoor effort, suggesting that more effort was needed. Stander (1998) found that an effort of 30 spoor samples and a distance of 1200 km were needed to reach an asymptote in the Kalahari area, whereas Balme *et al.* (2009a) required a driven distance of 600 km in order to reach an asymptote. Houser *et al.* (2009) required an effort of 1296 km and 38 spoor counts in the wet

season, and 3636 km and 90 individual spoor counts in the dry season in order to reach an adequate sampling effort for cheetahs in Botswana. Therefore, initial effort (transect lengths) should be based on a high penetration rate in one's sample area and subsequent sampling effort will rely on the number of spoor encountered. Although asymptotes were not reached, effort expended in this spoor count survey was sufficient to meet the desired levels of precision in terms of sampling intensity (Stander 1998, Balme *et al.* 2009a). These levels of precision allow the study to be repeated and this would be valuable in order to calculate a specific calibration equation for the Mangwe area in the future.

Funston *et al*.'s (2010) calibration equation

Funston *et al*.'s (2010) combined (all carnivore) model, calibrated for sandy soils, resulted in the lowest leopard density estimate in my study. Funston *et al*. (2010) found a significant linear relationship between known carnivore densities and track densities, sampled from a number of different areas with sandy soils. Because this equation is calibrated for a combination of carnivores (of which leopards represented the lowest sample size), I do not think this equation is suitable, nor the resulting density estimate of 1.28 leopards/100 km^2 accurate, for my study.

Stander's (1998) calibration equation

Stander's (1998) equation was based on known leopard densities in a semi-arid savanna woodland area of Namibia, where leopard densities were relatively low (~1 leopard/100 km^2). When Stander's (1998) spoor counts were calibrated with true density (1.45 leopards/100 km^2) the resulting density estimate (1.37 leopards/100 km^2) was very close to true density. Although Stander's (1998) calibration may be applied to the Mangwe area due to similar climate and habitats (semi-arid woodland savanna), the equation was calibrated specifically for leopards in a low-density area. The leopards in Stander's (1998) study area have some of the largest recorded home ranges (183 –   194 km^2 for females, and 210 – 1164 km^2 for males) (Stander *et al.* 1997) and the lowest densities, which are not reflected in the home range estimates of my study (Chapter 4) nor in the camera-trapping density estimates. Therefore, this equation is unlikely to be suitable for leopard density estimation in the Mangwe area.

Balme *et al*.'s (2009a) calibration equation

Balme *et al*.'s (2009a) Phinda calibration was based on a linear, but insignificant, relationship between spoor density and true density, and underestimated the density of leopards at Phinda (Balme *et al.* 2009a). The Phinda area is protected as a private game reserve, with no trophy hunting or poaching occurring within the reserve. However, the leopards within the reserve may serve as a source population for trophy hunting areas on the reserve's periphery, and population dynamics within the reserve are affected by this (Hunter & Balme 2004). Density estimates from Phinda, are similar to those using the same methods in this study (Balme *et al.* 2009a). Due to

less hunting pressure within the reserve, leopards are often sighted walking along the roads (Balme *et al.* 2009a); and therefore the spoor encounter rate may be higher than in the Mangwe area, which has no formal protection and is impacted directly by trophy hunting, poaching and PAC (Grant 2008). However, although the Phinda calibration resulted in an insignificant relationship, this equation did generate what appears to be a realistic density estimate (3.29 leopards/100 km^2) as this estimate is similar to the mean estimated camera-trapping densities in my study.

Recommended spoor density estimate

Therefore, the spoor density estimate of 3.29 leopards/ 100 km^2 in my study, calibrated using Balme *et al.*'s (2009a) equation is likely to be the most accurate population density estimate based on the spoor indices for the study area. Ideally, the spoor density estimate resulting from my survey should be calibrated with a true density estimate (Stander 1998, Funston *et al.* 2010). However, the cost and requirements involved in identifying individual leopards by their tracks with any certainty, in order to replicate these methods, were prohibitive (Stander 1998, Balme *et al.* 2009a). A number of surveys would be required in order to generate a calibration equation based on the camera densities for my study area, and this was not within the scope of the study.

Variables affecting spoor count indices

The variables that affect this method, such as substrate type, species behaviour (e.g. range utilization or road use), weather (e.g. excessive wind or precipitation), level of skill of personnel, and road traffic, may influence spoor detection in an area, and ultimately, the spoor frequencies and resulting density estimates (Stander 1998, Silveira *et al.* 2003, Balme *et al.* 2009a, Funston *et al.* 2010). Balme *et al.* (2009a) found that male leopards use roads more regularly than females at Phinda. In addition, males generally range further afield than females (Bailey 1993, Bothma *et al.* 1997, Stander 1997) making them more susceptible to being detected. This does not appear to have been the case in my study as only six of the 27 sets of spoor found were male. Individuals were not identified, and therefore, the frequency of spoor detection may be an indication of leopard density and the male: female density ratio in the area.

One aspect of species, or even individual behaviour, that is likely to have had a strong influence on the spoor frequencies in this study was the exposure of these leopards to human encroachment, trophy hunting and PAC (Kolowski & Holekamp 2006, Grant 2008). When compared to areas such as the Kruger National Park or Phinda Private Game Reserve, where there are no village settlements or trophy hunting, the leopards in these protected areas are regularly detected using the road network (Bailey 1993, Balme *et al.* 2009a). Although leopard spoor was detected on the roads in my study, higher detections of spoor were encountered

opportunistically when walking along the dry river beds of the area. It is also likely that the leopards of the Mangwe area are more likely to use the dense network of kopjes in order to navigate their territories, as these sprawling expanses of rock provide an easy route where leopards can travel undetected by humans, rather than using the limited road network in the area (Grant 2008). This behaviour could explain the lack of tracks detected in the northern section of the study area. Even though leopards were detected by the baited camera-traps placed in the north, no tracks were observed in this part of the study area. Leopards in the northern area may be more wary of using roads as there are three villages in the area, which presumably results in increased human activity. Consequently, leopard behaviour and the heterogeneous landscape could have decreased the incidence of spoor along the roads.

Another factor that is likely to have influenced the detection of spoor during my survey was the high number of cattle that used the roads. This usually occurred at around 10h00 every morning on transects number five and one. To try and circumvent this issue, transects number five and one were sampled first thing in the morning. However, on three occasions cattle were let out of their kraals earlier than this (or they had not been kraaled the previous night) and this reduced spoor visibility.

3.5.2 Camera- trapping density estimates

Contrary to the field limitations of detecting spoor, camera-trapping appeared to be a successful, efficient and non-intrusive method for surveying the leopard population of the Mangwe district, as reported previously (Karanth & Nichols 1998, Silveira *et al*. 2003, Jackson *et al*. 2005, Balme *et al*. 2009a). Camera-trapping protocols rely on a rigorous sampling design based on previous expert studies and a robust statistical framework (Otis *et al*. 1978, White *et al*. 1982, Pollock *et al*. 1990, Nichols 1992, Karanth & Nichols 1998, Borchers & Efford 2008, Royle & Young 2008, Royle *et al*. 2009). Although the unbaited camera-trapping survey was based on this framework, the slow shutter speed of the cameras used did not produce adequate results for a mark-recapture analysis of the leopards. This survey did, however, serve as a useful pilot study and produced some valuable data on the species present in the area, which could serve as a species inventory and an indicator of prey abundance (O'Brien *et al*. 2003, Mackenzie & Nichols 2004, Stein *et al*. 2008, Tobler *et al*. 2008).

Density estimates obtained during the baited camera-trapping survey varied considerably, depending on the buffer method used, as well as the software programme used to analyse the data. The highest density (7.04 leopards/100 km^2) was obtained from CAPTURE's M_h model using the HMMDM buffer (2.5 km) which resulted in an effectively sampled area of 313 km^2.

The lowest density estimate (1.37 leopards/100 km^2) was calculated by SPACECAP, using a buffer of 12 km to create a 'state space' area of 1237 km^2.

CAPTURE

The model selection algorithm of CAPTURE identified M_h as the most appropriate model, followed by the model M_o. Model M_o is thought to be too simplistic for free-ranging animals (Jackson *et al*. 2005) because of its underlying assumptions. This model assumes homogeneous capture probabilities throughout the survey period, therefore, that there are no behavioural responses to capture incidences and no variation in the experimental situation over time (Otis *et al*. 1978). These assumptions are usually unrealistic, and estimators of population size based on these are known to be sensitive to violations thereof (Otis *et al*. 1978, Karanth & Nichols 1998). However, the M_o model is useful in testing for sources of variation and providing a basic model upon which to base generalisations about capture probabilities (Otis *et al*. 1978, White *et al*. 1982). However, the M_h model and its jacknife estimator are thought to be the more robust of the two models (Otis *et al*. 1978, Karanth & Nichols 1998).

The M_h model is selected in most capture-recapture camera-trapping surveys (Karanth & Nichols 1998, Kawanishi & Sunquist 2004, Soisalo & Cavalcanti 2006, Balme *et al*. 2009a, Wang & Macdonald 2009) as the model allows for variation of capture probabilities among individuals, whilst each individual's probability of being recaptured remains constant over the sampling period (Karanth & Nichols 1998, Soisalo & Cavalcanti 2006). One would expect that leopard capture probabilities would be heterogeneous among individuals, due to age and sex specific behaviour (Otis *et al*. 1978, Trolle & Kery 2003, Wang & Macdonald 2009), and therefore this model fits the data and expectations well. However, one of the assumptions of the M_h model is that there is no behavioural response to capture (Otis *et al*. 1978). This assumption is likely to be violated by any camera-trapping survey that employs baits or lures to attract animals to the traps, or where the study animal has any favourable or unfavourable experience at the trap (Otis *et al*. 1978). Even flashes from the camera-traps could cause a behavioural response. Although this assumption was almost certainly violated in my study, the M_b model (which would account for behavioural responses) was not selected by CAPTURE (in fact, M_b was only sixth on the list of suitable models) and the M_h model still appeared to fit the baited camera survey data best. The jacknife estimator, which is associated with the M_h model, is the most robust technique for analysing a large number of occasions with a substantial recapture rate, which was the case in my study (Otis *et al*. 1978, Karanth & Nichols 1998, Soisalo & Cavalcanti 2006). Based on these arguments, of all the resulting CAPTURE density estimates the M_h model is likely to have produced the most reliable density estimates (Karanth & Nichols 1998) for leopards in the

Mangwe area. However, these were influenced by the buffer methods used, resulting in a range from 4.79±0.83 leopards/100 km^2 (MMDMOSA) to 7.04±1.22 leopards/100 km^2 (HMMDM).

CAPTURE's statistical tests indicated a lack of population closure for the baited camera survey. This may be one of the persistent limitations of capture-recapture models' assumptions (Soisalo & Cavalcanti 2006). The main assumption of a closed population is that the size of the population surveyed is constant over the investigation period (Otis *et al.* 1978). This means that there is geographic closure, facilitated by a boundary that limits the population; and demographic closure, meaning no births, immigration to, deaths, or emigration from the study area (White *et al.* 1982). This is a strong assumption that is very rarely true for biological populations, therefore it is generalised to mean that there are no unknown changes to the initial study population, and that any known losses do not violate this definition of population closure (Otis *et al.* 1978, White *et al.* 1982). One way of satisfying this assumption has been to limit the duration of the survey period to suit the longevity of one's study animal (Otis *et al.* 1978). For large carnivores which are relatively long-lived, a survey of up to three months has been considered a reasonable time-frame to assume population closure (Karanth & Nichols 2000). Importantly, previous work has used similar time frames: 59 days for jaguars (Silver *et al.* 2004) and 65-70 days for snow leopards (Jackson *et al.* 2006). Wang & Macdonald (2009) surveyed a population of tigers and leopards for 250 days without violating population closure. However, this was probably achieved by dividing their study area into five trapping zones which were sampled for 50 days each. Thus, my 66-day study should have resulted in a closed population according to CAPTURE's tests. However, even if population closure was in fact met, these tests could possibly have been reacting to the behavioural changes in the capture probabilities of the leopards, which the programme may have perceived as recruitment (Soisalo & Cavalcanti 2006). The attraction to the bait at the camera sites (i.e. trap-happy individuals) may have also resulted in a lack of population closure (Otis *et al.* 1978).

Buffer methods

The main requirement when estimating the size of the buffer is to determine how far individuals move outside the sampled area during the survey period (Otis *et al.* 1978, Balme *et al.* 2009a).The HMMDM buffer method has been found to underestimate the effectively sampled area by using comparatively short proxy distances for the buffer area. These are estimated from the MMDM between camera-traps, as opposed to the slightly longer and more accurate MMDMOSA, estimated from an abundance of GPS points from telemetry data (Soisalo & Cavalcanti 2006). This underestimation of the sampled area results in an overestimate of animal density (Soisalo & Cavalcanti 2006, Dillon & Kelly 2008). The HMMDM buffer is usually used

when home range data are lacking and it is used as a proxy for the home range radius (Karanth & Nichols 2002, Dillon & Kelly 2008). The density estimates calculated using HMMDM were the highest out of all the methods (5.63±0.15 and 7.04±1.22 leopards/100km^2) and could be overestimates (Soisalo & Cavalcanti 2006, Dillon & Kelly 2008). However, Balme *et al*. (2009a) did not find this method to overestimate densities in Phinda and found it to be the second best performing method, after the telemetry derived buffer of MMDMOSA.

The variation in buffer influence experienced in camera-trapping studies may result from influences such as the sample sizes of the animals captured twice or more by the cameras and the distance between camera sites (grid size). A larger sample size would give a more accurate reflection of the distance moved by the study species, and a camera-grid that is too small, may result in underestimates, as it does not accurately capture maximum distances moved (Dillon & Kelly 2008). The sample size of collared animals and the sex ratio of the sample will also affect these distances, as more collared animals in the study will result in more realistic movement data of the study animals, and uneven sex ratios may bias these distances, as males are known to travel further than females (Hamilton 1976, Bailey 1993, Mizutani & Jewel 1998, Swanepoel 2008, Balme *et al*. 2009a). The species' behaviour is also an important consideration, as the range size of a species will influence the size of the trapping grid as well as the number of cameras needed in order to recapture enough individuals in the study for a robust capture-recapture analysis to be done. Other studies report that the HMMDM buffer works best when the species' range size is small compared to the trapping grid area (Nichols & Conroy 1996, Dillion & Kelly 2008). Different individuals of the same species also have different home range sizes and movement patterns, therefore the size of trapping grids, camera spacing and buffer distances should perhaps be tailored to suit the local population (Dillon & Kelly 2008). It appears that different buffer methods suit different studies, based on a combination of these reasons; however, a standardised, detailed method of buffer estimation is needed in order to compare studies of the same species (Soisalo & Cavalcanti 2006, Dillon & Kelly 2008).

The MMDMOSA estimates (3.83±0.1 and 4.79±0.83 leopards/100 km^2), which were based on the actual distances travelled by collared leopards in my study, were lower than those derived from the HMMDM because of the larger buffer method (and hence, larger effectively sampled area). Balme *et al*. (2009a) warn that the sample size of the collared animals must be taken into consideration when using estimates based on the MMDMOSA method; when a low proportion of the population is collared, it may be risky to extrapolate such home range estimates to the entire population (Balme *et al*. 2009a). Their estimate of MMDMOSA was based on a large proportion (93%) of collared individuals in Phinda (Balme 2009). Only three leopards were

collared in this study (Chapter 4) and therefore the MMDMOSA was based on a small sample size. However, in the absence of any alternatives, these data represent a valuable insight into the movement patterns of leopards in the Mangwe area. The MMDMOSA density estimate (4.79±0.83 leopards/100 km^2) using the M_h model is likely to be more accurate than that of the HMMDM, and therefore the most reliable density estimate from the camera-trapping data.

SPACECAP

The new SECR programme SPACECAP is thought to be more robust than CAPTURE due to its incorporation of the spatial aspect of trapping data and the elimination of the *ad hoc* approach to estimating buffer distances (Singh *et al.* 2010). However, I found the estimation of the boundary strip, required in SPACECAP for the creation of the 'minimum area rectangle', to be equally as arbitrary as the methods of buffer estimation used in previous methods. Singh *et al.* (2010) state that population size will be sensitive to the extent of the state-space, but that the density estimates are not influenced by changes to the size of the state-space. Consequently, the state-space should be sufficiently large to ensure stability of the density estimate (Singh *et al.* 2010). However, the varying buffer distances and resulting state-space areas did have a significant influence on the population sizes and the density estimates obtained in my survey. A buffer of 12 km resulted in the lowest density estimate of all methods used (1.37±0.23 leopards/100 km^2) due to the large state-space created of 1237 km^2; a smaller buffer of 2.5 km resulted in a state-space (303 km^2) similar to the effectively sampled area created by the MMDMOSA buffer of 313 km^2. Therefore, of the SPACECAP density estimates, the estimate of 5.12±0.62 leopards/100 km^2 is likely to be the most accurate, considering the size of the sampled area estimate.

Despite the *ad hoc* 'state-space' estimation, SPACECAP does have aspects that make it a superior and more robust programme for estimating leopard density. Not only is the locality information of trap locations an important improvement, but by incorporating trap deployment details, a more accurate estimate of trapping effort can be obtained (Singh *et al.* 2010). One can therefore specify when certain trap stations were not operational during the survey, perhaps due to battery failure, theft or damage (Singh *et al.* 2010). Another important feature of the potential home range centres input file is that one can exclude areas that are unsuitable for the study species (Singh *et al.* 2010) and remove them from the estimated state-space (Royle *et al.* 2009). This is particularly useful in heterogeneous areas such as the Mangwe area, where villages, cultivated fields and large water bodies are present.

SPACECAP also allows a 'trap response' model to be analysed once all input files have been entered. This was especially important for my study, as it was assumed that the baits at the

camera sites would elicit 'trap-happy' behaviour due to the increased likelihood of a leopard returning to the trap (White *et al.* 1982, Soisalo & Cavalcanti 2006). The probability of captures estimated by SPACECAP in this study indicated a higher probability of capture after the initial captures and a high probability of capture overall. These probabilities of capture are higher than those estimated in other surveys that did not use baits. For example, 0.11 – 0.26 (Karanth & Nichols 1998), 0.03 – 0.07 (Silver *et al.* 2004) and 0.14 (Balme *et al.* 2009a).

Variables affecting baited camera-trap surveys

Baiting camera stations may therefore introduce bias. However, Henschel & Ray (2003) suggest that as long as the effort and the baiting protocol is standardised, the use of such attractants should not have any major influence on the data and does not pose any statistical problems for capture-recapture estimates. The justification for baiting in this study was due to the slow shutter speed of the cameras, resulting in poor capture results in the initial unbaited camera survey. Karanth & Nichols (2000) suggest that the single most important aspect in camera-trapping is to get as many photo captures as possible. Baiting the camera stations definitely increased the photo captures of leopards in my study. Of course, baiting camera-traps is not recommended for all areas and may not be feasible for some studies. It could be argued that, baiting may habituate leopards to baits, which could result in adverse effects such as poachers poisoning baits or leopards becoming reliant on these sources of food (Lopez-Bao *et al.* 2010). Some may even argue that leopards will acquire a 'taste' for cattle foetuses (if this is the bait used) which may increase depredation on livestock (Kruuk & Turner 1967, Nowell & Jackson 1996, Ogada *et al.* 2003).

Although these scenarios are possible, the relatively short survey period of my study was unlikely to condition leopards and cause them to become reliant on this supplementary source of food (Lopez-Bao *et al.* 2010). It has been reported that if natural prey are available in adequate numbers, leopards will hunt these in preference to livestock (Norton *et al.* 1986, Bowler 1991), which may be more of a risk for leopards to hunt for various reasons, such as the livestock's proximity to humans and open habitats. Thus, it is unlikely that leopards in my study acquired a 'taste' for cattle from the bait foetuses used. It could be argued that baiting attracted leopards from outside the survey area and artificially inflated my density estimates (Otis *et al.* 1978). However, because leopards are territorial (Bothma & Walker 1999), it is likely that the resident animals would have prevented neighbouring individuals from encroaching into their home ranges. The buffer systems created for the effectively sampled area would have also excluded any leopards whose home ranges occurred outside of the study area. Not only did baiting increase the number of leopard captures in this study, but it also allowed the cameras to be

moved off the roads where anthropogenic-related captures were higher. Ideally, a baited camera survey would not be necessary if the cameras had faster trigger speeds. However, if one's study animal is individually identifiable, a baited study reduces the amount of effort required by dramatically increasing recapture probability.

Recommended camera-trapping densities

Based on the above arguments, I would recommend a range of 4-6 leopards/ 100 km^2 for the Mangwe area, based on the results of the CAPTURE estimate using the MMDMOSA buffer and the SPACECAP estimate using a buffer of 2.5 km, respectively. This range includes the densities of male and female leopards combined and this should be kept in mind when extrapolating densities for off-take quotas. I would therefore further recommend using the lower density estimate when extrapolating for density estimates for the entire Mangwe region (See Chapter 5 for more detail). These density estimates are comparable to Balme *et al.*'s (2009a) and Chapman and Balme's (2010) research in Kwa-Zulu Natal where they used the same buffer methods, and are only slightly less than those estimated by Steyn (2007) using the same methods in the Tuli area of Botswana. However, my density estimates are significantly lower than the density estimates for leopards in India (Chauhan *et al.* 2005, Edgaonkar 2008) and those published by Smith (1977) from the Matopos National Park (17 leopards/100 km^2). Thus, my results are similar to those from other southern African studies which have used similar estimation methods, and which have similar heterogeneous bushveld habitats.

3.5.3 Conclusion

The underlying assumptions of the spoor-tracking method, and the tracking conditions required to accurately estimate densities (Smallwood & Fitzhugh 1995, Stander 1998, Karanth *et al.* 2010) make this method less attractive when compared to the data-rich camera-trapping approach. However, if an accurate calibration equation is calculated for a specific area, spoor counts are precise and therefore repeatable, and could be useful in the long-term monitoring of a leopard population in an area. Camera-traps can be placed in a multitude of habitats, are accurate in species determinations, and offer the possibility of identifying the age and sex of species, which is not always possible for spoor counts (Silveira *et al.* 2003). The results of this study have produced the first leopard density estimates based on camera-trapping for Zimbabwe, and the first leopard spoor index for the Mangwe area. In addition, I have shown that baited camera-trapping is a successful method for estimating leopard population densities in a non-protected, cattle farming area.

CHAPTER 4
HOME RANGE AND MOVEMENT PATTERNS

Collared male (M1) recovering from anaesthesia.

4.1 Introduction

Home range has been described as the area over which an animal normally travels in pursuit of routine activities, such as searching for food, caring for young, and mating (Burt 1943, Jewell 1966, Gittleman & Harvey 1982). However, this definition excludes any exploratory 'forays' and is therefore not the entire area that an animal covers during the course of its lifetime (Burt 1943, Jewell 1966, Bothma *et al.* 1997, Grimbeek 1992). A predator's spatial utilisation reflects its relationship with resources and other ecological factors, such as cover for hunting and concealment purposes, prey distribution and abundance, human influences, access to water and possible mates (Bailey 1993, Marker & Dickman 2005a). Information pertaining to these factors can provide an indication of the importance, distribution, and abundance of such resources and their contribution to overall behaviour patterns (Henschel 1986).

Leopards are generally described as solitary felids, with the only groups formed being those of a female and her dependent offspring, and between males and females who associate briefly to mate (Bailey 1993, Bothma & Walker 1999, Sunquist & Sunquist 2002). However, the non-random patterns of overlap of individual home ranges is sufficient to suggest that some form of social congruency exists among felid species; this has been classified as 'spatial groups' (Macdonald *et al.* 2010b). These spatial groups, or the spatial organisation within the groups, can be characterised by the extent of overlap within and between the home ranges of females and males, adults and sub-adults (Bailey 1993, Macdonald *et al.* 2010b). The socio-spatial organisation of these home ranges is determined by the territoriality of the leopards. Territoriality has been defined as any behaviour by an animal that tends to confine the movements of that animal to a particular locality (Etkin 1967). Bailey (1993) characterised the spatial organisation of leopards in the Kruger National Park by investigating the degree of overlap among adult males, adult females and transients. He found that there was little overlap between neighbouring resident adult males' home ranges. However, usually three or four, and occasionally up to six, females' smaller home ranges overlay a single adult male's range; and superimposed upon this were transient leopards (sub-adult or old adults) (Bailey 1993, Macdonald *et al.* 2010b). As with most felids, the spacing patterns of females are dictated by food supply and high-quality habitats in order to raise their young successfully (Bailey 1993, Mizutani & Jewell 1998). By contrast, the primary requirements associated with the home ranges of adult males seems to be access to, and successful breeding with a number of females, without interference from neighbouring males (Bailey 1993, Mizutani & Jewell 1998). This socio-spatial behaviour is an important factor to investigate, particularly in areas where leopards are persecuted. Male leopards practice infanticide, where an incoming male that takes over the territory of the current dominant male will kill any leopard cubs currently in the area, in order to

ensure the breeding of his own offspring (Bailey 1993, Scott & Scott 2003). If males are continually persecuted in an area, this creates a situation where females are not able to rear cubs successfully because of the high rate of male turnover and incursions (Balme *et al.* 2010a). Therefore, the home range sizes and ranging patterns of leopards are important factors in any conservation management planning, especially those concerning off-take quotas.

Relative to their metabolic needs, large carnivores which often predate on large prey will have large home ranges and low population densities (McNab 1963, Gittleman & Harvey 1982, Macdonald *et al.* 2010b). This would explain why both male and female leopard home range sizes increase in arid areas, where prey and shelter may be scarce (Bothma *et al.* 1997), and decrease in the more mesic habitats provided by forests and woodlands (Bailey 1993, Odden & Wegge 2005, Simchareon *et al.* 2008) (Table 4.1). Other factors that may influence home range sizes through animal behaviour include intra-guild competition and whether leopards are persecuted in the region or not (Marker & Dickman 2005a). The absence of larger carnivores in an area could result in an increased density of less dominant carnivores (Creel *et al.* 2001). However, it is uncertain how this influences leopards because of their opportunistic diet and behaviour, making it possible for them to survive in areas where they overlap with larger carnivores (Karanth & Sunquist 2000, Marker & Dickman 2005a). Previous studies have shown that hunting, and subsequent removal of leopards, can result in the expansion of the home ranges of any remaining leopards and increased overlap of territories (Marker & Dickman 2005a). Importantly, Marker and Dickman (2005a) found that leopard densities in protected areas were five times higher than those of areas without any formal protection. Evidence shows that there is a rapid decline in predator population densities outside of protected areas (Woodroffe 2001), highlighting the importance of density and home range studies outside of protected areas which make up only 13% of the leopard's potential range in Africa (Martin & de Meulenaer 1988). Leopards are often persecuted in these unprotected areas (Butler 2000), and these is usually little information on how they utilise the landscape. Identifying areas of important resource use, as well as ranging patterns and demographics of leopards outside of protected areas, are important steps towards research-based conservation planning (Simcharoen *et al.* 2008).

Leopards have highly variable home ranges throughout their distribution (Chapter 1: Figure 1.1). Even within sub-Saharan Africa, home range sizes differ greatly (Table 4.1). Although the environmental and behavioural patterns described above do influence leopard home ranges, another important consideration when comparing studies is the home range estimation method used. The Minimum Convex Polygon (MCP) estimator is still used in home range estimations, albeit mainly for comparative purposes, despite general recognition of its weaknesses; such as its

sensitivity to the number of location fixes, sampling duration and autocorrelation (Harris *et al.* 1990, Swihart & Slade 1997, Seaman *et al.* 1999, Laver & Kelly 2008). The Kernel Utilisation Distribution (Kernel UD), also known as the Kernel Density Estimator, is the more robust, efficient and unbiased estimator recommended for home range studies (Worton 1989, Seaman & Powell 1996, Swihart & Slade 1997, Seaman *et al.* 1999, Börger *et al.* 2006, Laver & Kelly 2008). However, the Kernel UD also has its weaknesses when attempting to compare home ranges between different studies; the main weakness being the selection of a smoothing factor (also known as the bandwidth or h value) which varies in each study (Laver & Kelly 2008). The smoothing factor greatly influences home range estimates because it can be calculated in a number of ways, and depends on the telemetry data sample size (Seaman *et al.* 1999, Laver & Kelly 2008). Consequently, as much supplemental information on the methods used, and the environmental factors affecting the area should be mentioned in home range studies in order to make more meaningful comparisons, and to replicate studies (Marker & Dickman 2005a).

The only published home range data for leopards in Zimbabwe is that of Smith (1977), who estimated home ranges of leopards in the Matopos National Park to be between 10 and 19 km^2, based on spoor-tracking. As mentioned in Chapter 3, the density estimates of leopards in this national park are likely to be higher than other parts of the country because of the high density of kopjes, the high-quality habitat available and the protection afforded to the animals living within the national park (Smith 1977, ICOMOS 2003). These factors, and the outdated home range estimation method used by Smith (1977), may have resulted in a comparatively small and potentially underestimated home range size. This has significant implications for the setting of reliable and sustainable hunting quotas. Thus, robust home range estimates and a sound understanding of leopard spatial and habitat requirements is required, especially in areas outside of protected parks, reserves or conservancies that are impacted upon by cattle ranching, human settlement and hunting practices.

Table 4.1: Summary of published data on the home range (HR) sizes of leopards, showing location, method of live capture, demography of study leopards, duration of tracking and the home range estimation method used and the average home range for male and female leopards (if specified in the study).

Authors	Location	Habitat type	Protected Area	Live capture method	Number & sex of captures	Duration of tracking	HR size (MCP 100%) km²	HR size (MCP 95%) km²	HR size (Kernel 50%) km²	HR size (Kernel 95%) km²	Average HR km²	
											MCP	Kernel 95%
Hamilton 1976	Tsavo National Park, Kenya	Thornveld hills	Yes	Cage traps	8 males (7 adults, 1 sub-adult) 1 female	64 -554 days	M: 20 – 59; 10 (sub-adult) F: 14				M: 30.5 (excl. sub-adult) F: 14	
Smith 1977	Matopos National Park, Zimbabwe	Savanna woodland	Yes	Based on interpretation of spoor			10 – 19				18	
Norton & Lawson 1985	Stellenbosch, Western Cape, #SA	Fynbos/ plantation	No	Cage traps	1 male 1 female	12 months	M:388 F: 487				M: 388 F: 487	
Norton & Henley 1987	Cedarberg, Wilderness, Western Cape, SA	Mountain fynbos	No	Cage traps	3 males	12-13 months	40 - 60				M: 50	
Grimbeek 1992	Waterberg, Limpopo, SA	Mountain bushveld	No	Cage traps Jump trap Coil	1 male 1 female		M:303 F: 157				M: 303 F: 157	
Bailey 1993	Kruger National Park, Mpumalanga, SA	Bushveld	Yes	Cage traps	5 male 5 females	24 months	M: 28 - 76 F: 14.8 - 18				M: 52 F:16	

Bothma *et al.* 1997	Southern Kalahari, Kalahari Gemsbok National Park, Northern Cape, SA	Desert grassland	Yes	*N/S	3 males 5 females	10 – 33 months	M: 717 - 1803 F: 186 - 693			M: 1614 - 2750 F: 200 - 908	**M: 578 F: 291**	**M: 2182 F: 489**
Mizutani & Jewell 1998	Lolldaiga Hills Ranch, north-west of Mt Kenya, Kenya (200 km^2)	Scattered woodland grassland	No	N/S	5 males 6 females	30 months	M: 64 - 110 F: 10 - 31				**M: 72 F: 22**	
Marker & Dickman 2005a	Waterberg Plateau, North-central Namibia (18 000 km^2)	Thornveld savanna	No	Cage traps	6 males 5 females	5 – 31 months		M: 125 - 312 F: 52 - 394			**M: 229 F: 179**	
Odden & Wegge 2005	Royal Bardia National Park, Nepal	Forest	Yes	N/S	2 males 1 female	3- 25 months	M: 20; 50 F: 17			M: 47; 48 F: 17	**M: 35 F: 17**	
Martins & Martins 2006	Cedarberg Mountains, Western Cape, SA	Mountain fynbos	No	N/S	2 males	10 weeks & 7 months					***N/S: 235-600**	
Simchareon *et al.* 2008	Huai Kha Khaeng Wildlife Sanctuary, Thailand	Dry tropical forest	Yes	Cage traps (wooden)	3 males 7 females	13 – 40.5 months		M: 48 F: 14 - 43		M: 35; 56 F: 18 – 41	**M: 48 F: 21**	**M:46 F:31**
Swanepoel 2008	Waterberg Mountain Range, Limpopo, SA	Mountain bushveld	No	Cage traps	1 male 3 females	57 – 603 days	M: 289 F: 41 - 291	M: 245 F: 38 - 221	M: 52 F: 14-23	M: 214 F: 94 – 109	**M: 289 F: 164**	**M: 214 F:88**

Balme 2009	Phinda Private Game Reserve, Kwa-Zulu Natal, SA	Lowveld bushveld and bushveld-grassland	Yes	Free darting Soft-hold foot snares Cage traps	14 N/S	12 months	36				**36**	
McManus 2009	Baviaanskloof (B.) & Addo Elephant National Park (A.), Eastern Cape, SA	Thicket	B: No A: Yes	Cage traps Foot snares	B: 2 males 2 females A: 1 male 1 female	3 - 12 months	B: M:183 - 632 F:123 -149 A: M:213 F: 148		B: M: 33-17 F: 22-23 A: M: 11 F: 27	B: M: 150-498 F: 106-130 A: M: 86 F: 110	**M: 343** **F: 151**	**M: 244** **F: 116**
Steyn 2007	Northern Tuli Game Reserve, Botswana	Low-tree savanna and shrub thicket	Yes	Cage traps Free darting	1 male 6 females			M: 78 F: 22 - 86	M: 6 F: 3-8	M: 79 F: 23 – 50	**M: 78** **F: 47**	**M:79** **F:36**
Chase-Grey 2011	Western Soutpansberg, Limpopo, SA	Bushveld to forest	No	Cage traps	1 male 1 female	4 – 6 months		M: 514 F: 14		M: 160 F: 16	**M: 514** **F: 14**	**M: 160** **F:16**

*N/S – Not specified in the article referenced.

#SA – South Africa.

4.2 Objectives

The objective of this chapter was to capture and track three leopards (at least one adult male) within the 200-km^2 study area, in order to provide accurate home range and movement data for the personnel responsible for making conservation decisions for the Mangwe area.

4.3 Methods

The capture, collaring and tracking of leopards was conducted from June 2010 to November 2010. It was intended that six leopards were to be captured and collared, however, due to initial budget, and the resulting time constraints; only three leopards (one male and two females) were captured. The home range sizes, home range overlap and movement patterns for these animals were analysed.

4.3.1 Capture and immobilisation

To obtain the data on which to base home range estimates, the live capture of leopards was necessary in order to attach GPS collars. Large carnivores can be captured using a variety of techniques. Some of the techniques commonly used for leopard capture include cage trapping (Hamilton 1976, Norton & Henley 1987, Bailey 1993, Marker & Dickman 2005a), free-darting (Bertram 1982, Steyn 2007, Balme 2009) and soft-hold foot-snaring (Frank *et al.* 2003, Balme 2009, Sikes *et al.* 2011). Wild animals are more prone to stress and injury than domestic animals, especially during capture or restraint (Tribe & Speilman 1996). In addition, predators in general, and leopards in particular, are potentially dangerous to the personnel involved in any immobilisation procedure. Therefore, human safety must also be taken into consideration (Tribe & Spielman 1996, la Grange 2006, Balme *et al.* 2010a). The main aim of every capture procedure should be to minimise the stress on the animal, whilst maximising the safety of any personnel involved (Tribe & Spielman 1996, Sikes *et al.* 2011). This is achieved through thorough preparation and planning prior to the capture procedure, having a trained and experienced veterinarian on hand, limiting the personnel involved to as few people as possible in order to reduce noise levels, aiming to conduct the immobilisation procedure during the cooler parts of the day, and finally, monitoring the animal's recovery from a safe distance to ensure that it is not attacked by other predators, and that it is able to leave the capture site (Tribe & Spielman 1996, la Grange 2006).

For the current study, leopards were captured using baited cage-traps (Bailey 1993, la Grange 2006). Cage traps were chosen as the preferred capture method due to their relatively simple set-

up and operation procedures which required little skill. They were deemed to be comparatively safe to both personnel and the leopards, and they were kindly loaned to the project at no cost by the Dambari Wildlife Trust. The cage traps used were originally designed for cheetah capture, and therefore slight adjustments were required to make them appropriate for leopard capture. These cages had a double drop-door system; however for the current project purpose; one door was wired closed and the cage was used as a single drop-door trap (la Grange 2006). This door was triggered by a simple positive direct system mechanism, using fishing trace (100 kg breaking strain) which connected a nail-like trigger (Inset 3 of Figure 4.1) to a bait that hung within the trap. When the bait was moved, the mechanism was triggered by pulling out the nail from under the drop-door, thereby allowing it to slam closed. A locking mechanism was attached to the cage (Inset 4 of Figure 4.1), preventing the door from being slid open once it dropped closed (de Wet 1993). Bolts were also attached to the bottom of the door frame in order to create a gap of about 4 cm when it was closed (Inset 6 of Figure 4.1). This prevented the door from possibly slamming onto a leopard's tail (JR Peek pers. comm.). The cages' frames were made up of 20 mm square section tubular steel, enclosing 3 mm thick weld mesh grids of 60 mm x 25 mm and the dimensions of the cages were 308 cm x 80 cm x 90 cm. Cattle foetuses were used for bait, and these were hung approximately three-quarters of the way towards the back of the cage (Inset 5 of Figure 4.1) in order to lure the leopard as far into the cage as possible to prevent injury or escape as the trap door closed.

Prior to the capture phase of this study, cages were left out in the field with both doors wired open in order to rid them of any human or foreign scents. When possible, animal entrails (sourced from baits being used for trophy hunting in the area) or catfish (*Clarias gariepinus*) entrails were smeared over the cages, with the aim of enticing leopards to investigate the cages, without being captured. Cage trap sites were chosen based on the results of the baited camera-trapping photographs (see Chapter 3). Photographs were assessed to determine the sex and age of the leopards, as there were only three GPS collars available, and the aim was to collar at least one mature male. The baited camera sites served as a pre-baiting exercise to the cage traps (Hamilton 1976, Grant 2008), and when a suitable leopard started feeding at a particular camera site, the bait was then moved into the cage trap and the trap set.

1. Alternate door wired closed.

2. Single drop-door used, shown in the 'set', or open, position.

3. Trigger pin position.

4. Safety locking mechanism position.

5. Bait positioned towards the back of cage.

6. Stopper bolts positions.

7. Cage halves bolted and wired together.

90 cm

3. Trigger pin made from 10 cm mild steel nail. Loop soldered at the head to facilitate steel trace attachment. Pin placed through wire loop (attached to the cage) and positioned under the trap door. The length of nail under the door is adjusted for release sensitivity.

4. Spring-loaded safety locking mechanism, riveted onto the trap frame and positioned to block any upward movement of the door once closed.

5. Bait (cattle foetus) wired into position towards the back of the cage (near the closed door). Steel trace attached to the bait was threaded over the top of the cage and attached to the trigger pin, resulting in a direct positive release when the bait was moved.

6. Stopper bolts (1.2 cm) allowing a 4 cm gap to remain under the closed drop-door to prevent injury to a captured leopard's tail.

Figure 4.1: An example of the adapted cheetah cage traps used to capture the three leopards in this study. The dimensions of the cages were 308 cm x 80 cm x 90 cm. Although the cage pictured here functions as a double drop-door system, one door was wired closed and the cages were used as single drop-door traps. Minor alterations, as shown in insets 3, 4 and 6, were necessary in order to use the cages for leopard capture.

Traps were placed in densely vegetated areas to provide shade for captured leopards, whilst also camouflaging the cages. Where possible, the back of the cage was pushed up against a large rock face, or dense thicket, in order to create the feeling of a cave or dense thicket where leopards often cache their kills (Smith 1977, de Ruiter & Berger 2001, Swanepoel 2008). Cages were further camouflaged with thorn branches (*Ziziphus mucronata* and *Vichellia karroo*) around the sides and on top of the cage (taking care not to interfere with the trigger mechanism); this had the effect of restricting access to the bait and the sides of the cage, making the only access to the bait via the open door (la Grange 2006). The floor of the cage was covered with soil and leaf litter, mimicking the surrounding area, and preventing the leopard from detecting any wire underfoot. Two Dormicum Midazolam (30 mg) (Roche Laboratories, South Africa) tablets, a mild tranquiller, were also inserted into some of the baits set in the cages with the aim of calming any leopard whilst it was contained in the cage. Cages were set in the late afternoon and were checked first thing every morning (06h00). Once all preparations for setting the cages were completed, animal intestines were dragged around the area in order to spread the smell of the bait, in an attempt to cover any human scent. Capture effort was calculated as the total number of trap nights that were required to capture the three leopards, multiplied by the number of cages that were set on those nights.

All cages could be viewed from a safe distance (~40 m) with the use of binoculars (Nikon, 12 x 42 Monarch) to ascertain whether there was a leopard captured or not. Non-target species were released by standing on top of the cage, releasing the lock mechanism and pulling the door open. If a leopard was detected, the local veterinarian was alerted via satellite phone. Once the veterinarian had arrived, personnel involved in the capture approached the cage behind blankets in order to reduce stress on the captured leopard. These blankets were then placed over the cage to block the leopard's view and to help it relax (la Grange 2006), allowing the veterinarian to approach the cage for darting purposes without being detected. Blankets were used, as opposed to a tarpaulin, as they were more pliable and quieter when moved against the cages. The vet would then approach the cage and dart the leopard through a hole in the blanket. Once the dart had been administered, all personnel involved remained at distance of ~30 m away from the cage in order to minimise any disturbance. The cage was approached after about 15 minutes to determine recumbency.

Leopards were darted in the cage by a Dan-Inject® (Dan-Inject® International, Skukuza, South Africa) gas-powered dart gun on a gas setting of 2 - 2.5 bars. Dan-Inject® 3.0 ml darts were used, with a 30 mm long x 1.5 mm diameter collared needle. It is recommended that animals be

darted intramuscularly in any of the large muscle areas, preferably the rump (Deem & Karesh 2005, la Grange 2006, Balme *et al.* 2011) (Figure 4.2).
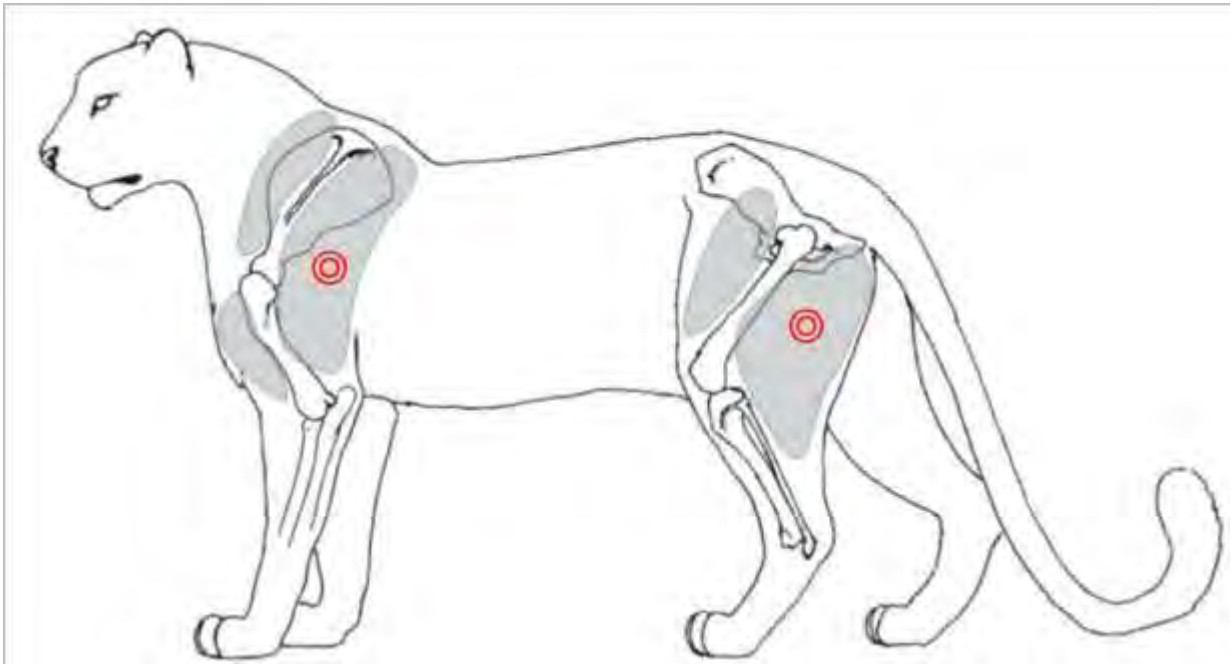
Figure 4.2: Recommended dart placement sites for felids (grey shaded areas), with preferred muscle area marked with ◎ (Adapted from Balme *et al.* 2011).

The drug used was Zoletil 100 (Tiletamine and Zolazepam, Virbac Animal Health, South Africa); the 2.5 ml solute was diluted at 200 mg/1 ml (JR Peek pers. comm.) and administered at dosage of 5 mg/kg of the leopard's estimated mass (la Grange 2006). Once the drug had taken full effect and the leopard was fully immobilised, it was removed from the cage, weighed, and then placed on a tarpaulin in the shade in a position that allowed it to breathe easily, usually in lateral recumbency (Deem & Karesh 2005). An ophthalmic ointment (Teargel, Adcock Ingram Ltd., Bryanston, South Africa) was used to prevent infection and drying out of the eyelids, which remained open whilst the animals were anaesthetised. A blindfold was placed over the eyes to prevent any damage from excessive exposure (la Grange 2006). As suggested by Deem & Karesh (2005), respiratory rate, temperature, and heart rate were monitored throughout the immobilisation procedure (Table 4.2).

Table 4.2: The normal physiological parameters and monitoring intervals that should be observed in the immobilisation procedures of free-ranging felids (Deem & Karesh 2005, Balme *et al.* 2011).

Physiological parameter	Normal range	Monitoring intervals
Temperature	37 – 39.5 °C	Every 10 minutes
Respiratory rate	8 – 24 breaths/minute	Every 5 minutes
Heart rate	70 – 140 beats/minute	Every 5 minutes

Whilst the physiological parameters were being monitored, the next most important objective was to attach the collar. Collars were tightened by leaving a gap large enough to slide one's hand under each collar whilst still fitting tight enough to prevent them being shed or moving around excessively (causing chafing) when the animal moved (Kenward 2001). Once this was done, morphological measurements were taken (Appendices VIII & IX), and the leopards were assessed for general health, breeding condition, injuries and parasites; they were also photographed. Blood smears and hair samples were collected, as well as a sample of ectoparasites which were stored in 70% ethanol. Leopards were aged, based on their size and body condition, but mainly by the colour, wear and eruption of their teeth (Bailey 1993, Stander 1997). Claws were checked for possible damage sustained whilst the leopard was contained in the cage, and treated with topical spray (Necrospray, Bayer Health Care, Isando, South Africa) if necessary. Any surface injuries were sprayed with Necrospray and open wounds were cleaned with hydrogen peroxide and treated with an antibiotic ointment (Curaclox DC, Norbrook Laboratories, UK) (Balme *et al.* 2011).

Once the leopards started showing signs of recovery, they were placed in the recumbent position in a shaded area which was monitored by the team from a safe distance, to prevent any disturbance or stimulation whilst the cat was regaining consciousness. The leopard was placed on soft undergrowth or soil, and any potentially harmful objects, such as sharp branches or rocks, were removed from the area to prevent the animal injuring itself when stumbling around, before it had fully recovered. The blindfold was left loosely in place as the leopards' pupils were still fully dilated and exposure to sunlight may cause damage to the retina (Seidensticker *et al.* 1974, JR Peek pers. comm.). Respiratory rate continued to be monitored from a distance by watching the thoracic expansions through a pair of binoculars (Balme *et al.* 2011). The leopards were monitored until they were fully recovered, and able to move away. Capture and handling procedures were conducted under the Rhodes University Ethical Standards Committee's ethical standards protocol, clearance number 2009Q1-5.

4.3.2 Telemetry equipment

All leopards were fitted with GPS/UHF-VHF collars (Africa Wildlife Tracking Cc (AWT), Pretoria, South Africa). These collars are made up of an on-board GPS system, with non-volatile memory and a VHF transmitter. The GPS transmitter records an animal's position as determined by the co-ordinates estimated by satellites. Collars recorded date and time, co-ordinates, altitude, velocity, temperature and positional dilution of precision (PDOP). PDOP is a measure of precision which is determined by the number and position of satellites used to record each positional fix. A PDOP value higher than five is rated as moderate to poor, and values of five or

less are rated as good to excellent (AWT). The data were stored on the collars and downloaded via UHF signal to a UHF-enabled console (HAWK-UHF, AWT) where the data were saved in Excel 2007 (Microsoft) spreadsheets. The collar fix data can also be downloaded via a 'download box', which is a remote receiver that can be left out in the field. The download box searches for any GPS collars within range (approximately 1 km) via the UHF signal. When a collar falls within this range, its data are stored on the download box (AWT). The data from the download box are then retrieved via the UHF-enabled console. The VHF transmitter is used to locate the leopards via a VHF receiver and antenna, this is necessary in order to get within a distance of < 1 km of the collar, to download its data via the UHF console.

The number of fixes that one sets the collars to record, and the upload interval scheduled, determine the battery life of the collars, therefore a balance is needed between an adequate number of fixes, and a long-lasting battery (see Appendix X which estimates the approximate battery life for a combination of different fixes and upload schedules). GPS collars were set to log GPS fixes six times a day at four-hourly intervals (02h00, 06h00, 10h00, 14h00, 18h00, and 22h00) (See the Autocorrelation section below for justification). Collar upload intervals (when the UHF console can communicate with the collars) were set to upload every five minutes from 05h00 to 18h00 every day.

4.3.3 Home range analysis

Minimum Convex Polygons (MCPs)

The MCP area method is the simplest home range estimation method used, as it measures the area of a convex polygon that links the outermost telemetry points (Mohr 1947). MCP is also the most widely used home range estimator, however it is generally used for comparative purposes only. The main drawback of this method is that the shape of the polygon, and therefore the home range area, is influenced by outlying fixes (excursions) as this may include areas that are not within the animal's 'normal' foraging range, and may also include large areas of unused space (Harris *et al.* 1990, Kenward 2001).

Once the data from each of the collars were formatted appropriately in Excel (Rodgers & Kie 2011), they were imported into ArcGIS 9.3.1 (ESRI, California). Home range tools extension (HRT) (Rodgers & Kie 2011) for ArcGIS was used to calculate MCPs. Both 95% and 100% polygons were calculated for comparison with other studies. The 'fixed mean' method option in HRT was selected, which calculates the mean of all co-ordinates, then selects the requested percentage of points closest to that mean point (Rodgers & Kie 2011). The number of GPS fixes

needed to estimate a robust home range size is an important factor (Kenward 2001, Börger *et al.* 2006). In order to ascertain whether the data reaches an asymptote, one has to plot the size of the MCP against the number of consecutive location fixes. This area is likely to increase rapidly with the number of initial fixes, and it is unlikely that these first points will represent the boundary of the home range of the study animal (Kenward 2001). As more locations are recorded, an asymptote is reached, after which new observations add little to the measured home range size (Kenward 2001). Beyond this asymptote, the home range may well continue to increase slightly, as the animal makes occasional excursions, but a reasonable estimate of home range has been obtained in the short term (Kenward 2001). Incremental analysis in Biotas™ 2.0 (Ecological Software Solutions, Florida, USA) was used to assess whether the MCP home range areas reached asymptotes when plotted against the number of GPS fixes used.

Kernel Utilisation Distribution (UD)

Previous studies have demonstrated that Kernel Utilisation Distribution (UD) methods provide more accurate and meaningful home range estimates than MCPs (Worton 1989, Seaman & Powell 1996, Swihart & Slade 1997, Seaman *et al.* 1999). Kernel UD is a probability density estimation, which provides an advantageous third dimension to home range estimates; and that is the amount of time that the animal spends in any area of its range (Worton 1989, Seaman *et al.* 1999, Rodgers & Kie 2011). The Kernel UD method places a kernel (a probability density) over each observation point (GPS fix); a regular grid is then superimposed on the data and an estimate of the density of all the overlapping kernels at that point is obtained for each grid intersection (Seaman & Powell 1996, Rodgers & Kie 2011). A kernel probability density estimator (UD) is then calculated over the entire grid using these probability density estimates from each intersection. Thus, the density estimate will be high in areas with many fixes and low in areas with fewer fixes (Seaman & Powell 1996). Home range estimates are then derived by contour lines (isopleths) that are drawn, based on the summed volumes of the kernels at the grid intersections. These isopleths define home range polygons at different probability levels and the area of these polygons are then calculated (Rodgers & Kie 2011). Home range tools (HRT) (Rodgers & Kie 2011) extension for ArcGIS was used to calculate 50% and 95% isopleths for Kernel UD in this study. The 95% Kernel UD was determined as the maximum general range for each leopard and the 50% Kernel UDs were the core areas or activity radii for the leopards (Seaman & Powell 1996).

Smoothing factor

A key issue when estimating Kernel UDs is determining the width of the kernels. This width is referred to as the 'bandwidth', 'smoothing parameter/factor', 'window width' or 'H value'

(Seaman & Powell 1996). Narrow kernels allow nearby fixes to have the greatest influence on the density estimate, thus revealing the small-scale detail of home ranges based on only a few observations. Wider kernels allow distant fixes to have more influence, thus revealing the general extent of the animal's distribution which may obscure the fine detail required to identify centres of activity (Seaman & Powell 1996, Rodgers & Kie 2011). The choice of smoothing factor can greatly influence home range estimates and therefore an optimal smoothing factor is of utmost importance in Kernel UD estimations (Rodgers & Kie 2011). There is still no agreement in the literature on how to best estimate this smoothing factor. However, the HRT extension in ArcGIS provides several automated and subjective methods for determining the optimal smoothing factor for spatial data (Rodgers & Kie 2011). Rodgers & Kie (2011) suggest that the optimal method of estimating smoothing factor with HRT is to use the automated value, referred to as the h_{ref} and to incrementally decrease (or increase) its proportion, until the outermost isopleth breaks down (or becomes a continuous line) (Rodgers & Kie 2011). The optimal h_{ref} would then be at the point, or proportion, where the isopleth becomes a continuous line, as opposed to a 'clumped' or broken-down isopleth. Although not fully automated, this process is repeatable and therefore valid (Rodgers & Kie 2011). In this study, I used the automated method in HRT to select a 'starting value' for the smoothing factor and then increased and/or decreased this h_{ref} value until a continuous isopleth resulted for the 95% UD. This value is unique and corresponds to each individual leopard's GPS fixes.

Autocorrelation

Autocorrelation and the number of fixes required to robustly estimate home range size have been debated extensively in the telemetry literature (Swihart & Slade 1985, Seaman & Powell 1996, Seaman *et al.* 1999, Solla *et al.* 1999, Börger *et al.* 2006). Autocorrelation is the result of the violation of the statistical assumption of independence between pairs of observed points. As such, it is the lack of independence between sequential observations at specific distances in time or space (Swihart & Slade 1985, Legendre 1993). Because animals typically move in a non-random manner, animal telemetry data are often strongly autocorrelated, particularly when frequent observation fixes have been collected (Solla *et al.* 1999). Swihart and Slade (1985) argue that frequent monitoring of individuals severely jeopardises the validity of the independence assumption, and recommend restricting the sampling regime based on results from pilot studies. Another more common method of eliminating autocorrelation has been to sub-sample datasets (Worton 1989). Recently, it has been argued that sub-sampling datasets not only reduces the sample size but also reduces the biological significance of the data (Solla *et al.* 1999, Börger *et al.* 2006). Also, an adequate sample size is deemed to be more important than independence between observations (Reynolds & Laundre 1990, Otis & White 1999). Thus, in

an attempt to reduce autocorrelation in this study, the scheduled GPS fixes of the collars were set to log at equal intervals, spaced four hours apart, resulting in a total of six fixes per day. These fixes were spaced evenly over 24 hours in order to monitor the leopards' diel movements. Despite being generally regarded as nocturnal, leopards are often active during the day (Bailey 1993, Bothma & Walker 1999). Thus, all GPS fixes logged by the collars were used in the home range analyses.

Home range overlap

The degree of overlap amongst the three collared leopards was measured as the area of overlap, as a proportion of each leopard's range (both MCP and Kernel UD). Coarse MCP estimates were also calculated for two other females (F3 and F4) that had been photographed at four or more camera-trapping sites. It was assumed that this information would provide some indication as to where other females within the study area ranged, in relation to the collared leopards. In the absence of any alternative home range data for males in the area, sites which produced photographs of other adult males were also plotted, in order to provide an indication of whether any of these males overlapped the collared male's home range.

4.3.4 Movement patterns

Distances moved between GPS fixes were calculated using the HRT extension in ArcGIS, which measured the straight line distances between fixes and was thus a minimum estimate (Merril & Mech 2003). Total distances moved per day were compared using a Kruskal-Wallis test (Statistica 10, Tulsa, OK, USA) because the assumption of normality was not met (Sokal & Rohlf 1995). The significance of pairwise differences between leopards was determined by multiple comparisons of mean ranks (Sokal & Rohlf 1995). Diel movements were investigated by measuring the distances moved between fixes at different times of the day. I tested the null hypothesis that distances moved at different times of the day were not significant; using a Friedman's Repeated Measures ANOVA (Statistica 10).

4.4 Results

4.4.1 Capture and immobilisation

Trap effort totalled 225 trap nights resulting in the capture of one male and two female leopards. This represents 75 trap nights per leopard captured. By-catch included four honey badgers (*Melivora capensis*) and two African civets (*Civettictis civetta*). Females F1 and F2 were fitted with GPS collars weighing 678 g each, representing 1.9% and 2.3% of their body mass, respectively. The male's GPS collar weighed 706 g and was 1.1% of his body mass. All collars

were below the recommended weight limit of less than 3% of an animal's body mass (Kenward 2001, Sikes *et al.* 2011).

The male leopard (M1) was captured on 19 July 2010 at site 17 (Figure 4.7). A camera-trap photograph of M1, sitting about two meters from the cage, was captured at 03h09 on the morning of his capture, giving some indication of the time he entered the cage. He was first darted at 10h14 with a dose of 475 mg of Zoletil in his left rump, followed by a top-up of 500 mg at 10h40 (Table 4.3). M1 remained calm when the darting team approached the cage behind the blankets, and only showed aggression by attacking the cage when the blankets were placed onto the cage, after this he calmed down and barely responded when darted. M1 weighed 62 kg and his estimated age was between five and six years. His morphological measurements are provided in Table 4.3 There was no discernible damage to any of M1's claws from the cage. However, there was some abrasion detected on his maxillary right canine (Figure 4.3.b), where the posterior edge (2) showed some green paint from the cage and some white enamel where the tooth had been damaged. It also appeared that a small piece of the tip of this canine had been chipped off in the cage (1). The left canines (Figure 4.3.a) show no damage, although some green paint from the cage was discernible on the mandibular canine (1), and there was some abrasion to the gum above the maxillary canine (2). M1 started showing signs of recovery by lifting his head at 13h08 and was fully recovered (able to walk away) at 15h40. M1 was tracked for just over eight weeks (57 days) (Table 4.4) until he was shot by a trophy hunter. An attempt was made to capture and collar another male leopard; however, after an additional 100 traps night without success, it was no longer feasible to continue trapping.

Table 4.3: Morphological measurements of the three captured leopards in the Mangwe area.

Measurements:	M1	F1	F2	Measurements (cm):	M1	F1	F2
Weight (kg)	62	35.5	29	Nose to tail tip	227	187	187
Age estimate (years)	5-6	7	2	Tail	77	67	84
				Chest girth	87.5	63	60.5
Times:				Neck girth	55	46.5	44.5
Dart-in time	10:14	10:57	09:19	Shoulder height (leg bent)	66	58	51.5
Initial recovery signs time	13:08	12:00	10:29	Shoulder height (leg straight)	78	62	60
Time at recovery	15:40	14:25	11:55	Skull width	17.3	13.5	12.6
				Skull length	27	22.1	20.3
Dosage:				*Canines:*			
Zoletil dart dosage (mg)	475	300	300	Top right	4	2.7	2.7
Top-up dosage (mg)	500	-	-	Bottom right	3	2	2.3
				Distance between top tips	5.5	4.5	3.6
				Distance between bottom tips	4.5	3.7	3.5
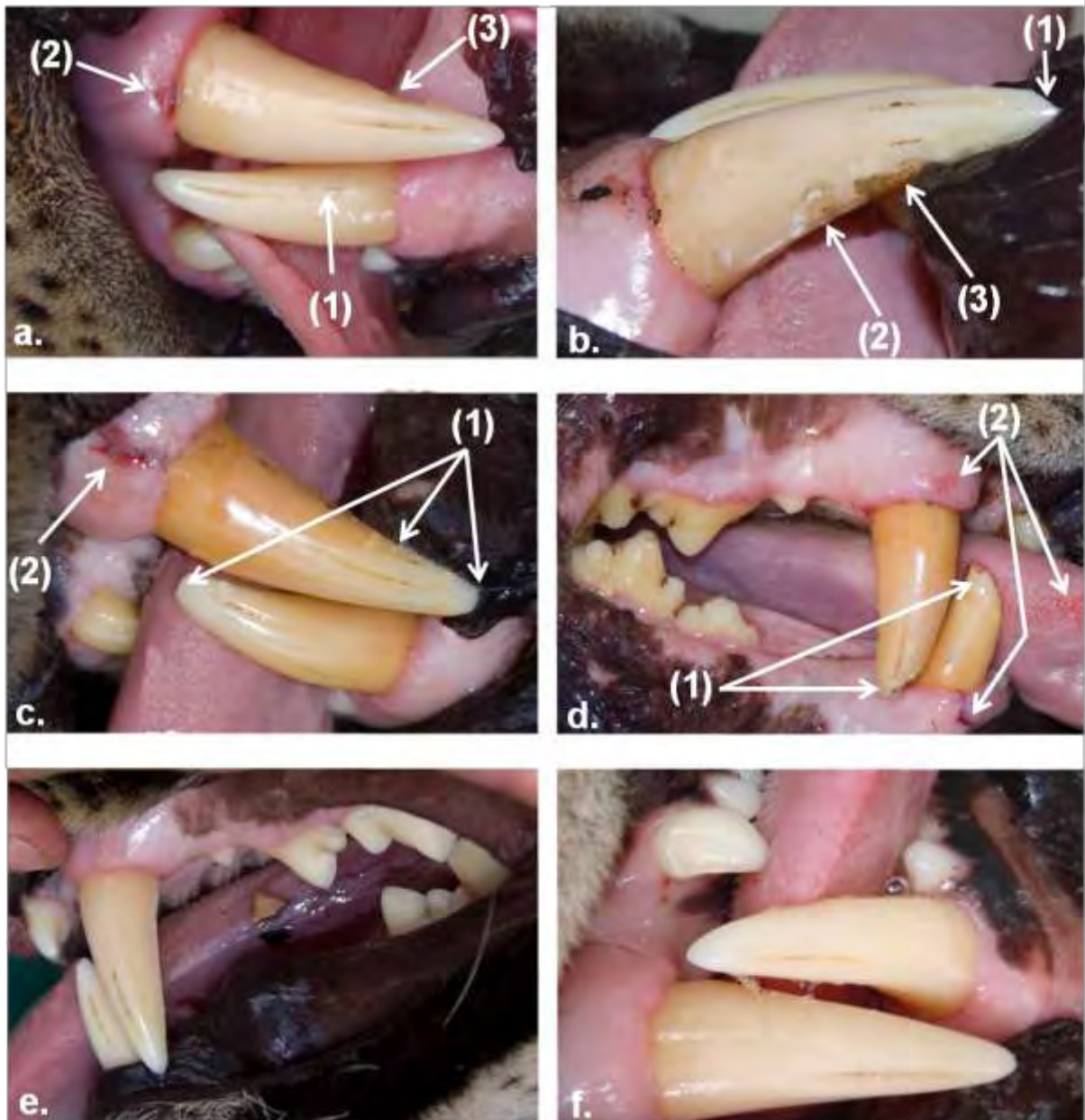
Figure 4.3: Photographs of the captured leopards' teeth, showing any damage inflicted whilst in the cage traps, or prior natural damage, and the varying colouration of the teeth – one of the parameters upon which the age estimates were based.

a. M1's left canines showing some detection of cage paint (1) and an abrasion on the gum (2), otherwise no new discernible damage; some older damage to the posterior edge of the canine can be seen (3).

b. M1's right canines showing the damage sustained from the cage capture, highlighting the broken tip of the canine (1) and some new damage to the enamel and the remains of some green paint from the cage on the posterior of the canine (2). Prior damage to the posterior surface can also be seen (3). Based on the natural wear and the slight yellowing of M1's teeth, his age was estimated to between five and six years.

c. F1's left canines show evidence of wear with age (1), the gums show some abrasion from the cage (2).

d. F1's right canines also show some old wear and breakages (1), but no recent damage except some abrasions on her tongue and gums (2). Based on the natural wear and severe yellowing of F1's teeth, her age was estimated to be seven years or older.

e. F2's left canines and molars showing no damage from the capture process.

f. F2's right canines, showing no damage from the capture process. Based on the lack of natural wear and lack of yellowing of F2's teeth, her age was estimated to be about two years.

The first female (F1) was captured on 5 August 2010 at site 15 (Figure 4.7). It was not possible to determine the time of F1's capture as the camera-trap placed at the capture site did not trigger before she entered the cage. F1 was darted in the left shoulder at 10h57 with 300 mg of Zoletil. No top-up doses were necessary. F1 weighed 35.5 kg and her age was estimated to be about seven years. Other morphological measurements are presented in Table 4.3. F1 appeared much more aggressive than the male when approached by the darting team, and continued to growl and attack the cage until the drug took effect. As a result of this aggressive behaviour, F1 sustained injury to five of her claws (three on the front right paw, one on the front left and one on the back right), some of

Figure 4.4: A photograph of F2's injured claw, showing how the nail is split but not broken.

which were split, but none were completely broken off (an example of a split claw is given in Figure 4.4). Despite the damage to F1's claws, little apparent damage could be detected on her teeth, although there were some abrasions on the gums. Most of the tooth wear and breakages appeared to be old (Figure 4.3.c & d).

F1 starting showing signs of recovery at about 12h00 and was fully recovered by 14h25. F1 was tracked for just over 15 weeks (107 days), the longest period of the three leopards (Table 4.4). The UHF signal from her collar stopped communicating with the download console (this was first noticed on the 7 November) and the collar was tracked via VHF to the same position for four subsequent days. It was assumed that the female was denning, and in order to avoid any dangerous encounters that could have resulted from investigating the area, a download box was erected at the site, which would connect to the collar and download the fix data if the female left the lair. When I returned to the field three months later, the collar was still in the same location, and the download box had not downloaded any data from the collar (this was because of the breakdown in the UHF receiver on the collar). After verifying the collar's location as stationary for four subsequent days, I decided to investigate, as I was concerned that the leopard may have died. Upon investigation, I found that the collar had been shed in a small crevice high up on the kopje. Even with the collar in hand the download console was still unable to detect the UHF

signal from the collar. The collar was sent to AWT where technicians were able to download the GPS data. The last movement data from F1 was recorded on the 20 November.

The second female (F2) was captured on the 25 August 2010 at site 5 (Figure 4.7) and was also not captured on camera beforehand. F2 was darted at 09h29 with a dose of 300 mg of Zoletil. No top-up dose was needed. F2 was relatively calm when approached by the darting team to cover the cage and growled and moved around in the cage when darted, but did not attack the cage. F2 weighed 29 kg, and her estimated age was about two years old. Other morphological measurements are presented in Table 4.3. F2 had two injured claws, but these injuries were minor, as the claws were slightly split in places, but not broken (similar to F1) (Figure 4.4). F2's teeth were all intact, with no discernible damage, new or old (Figure 4.3.e & f). F2 was tracked for 11 weeks (78 days), after which she could no longer be located with the VHF receiver. Assuming that either the VHF batteries on the collar were flat and/or that F2 had shed the collar, a camera-trap and bait were set up within her known core area in an attempt to photograph her. On 7 March it was confirmed that F2 had shed her collar, and I was unable to re-locate it with the VHF receiver. F2's collar was attached with a relatively larger gap (a whole hand space, as opposed to the three or four fingers used for M1 and F1) (Figure 4.5) because she was a sub-adult and we were concerned about the effects the collar would have on her as she grew.
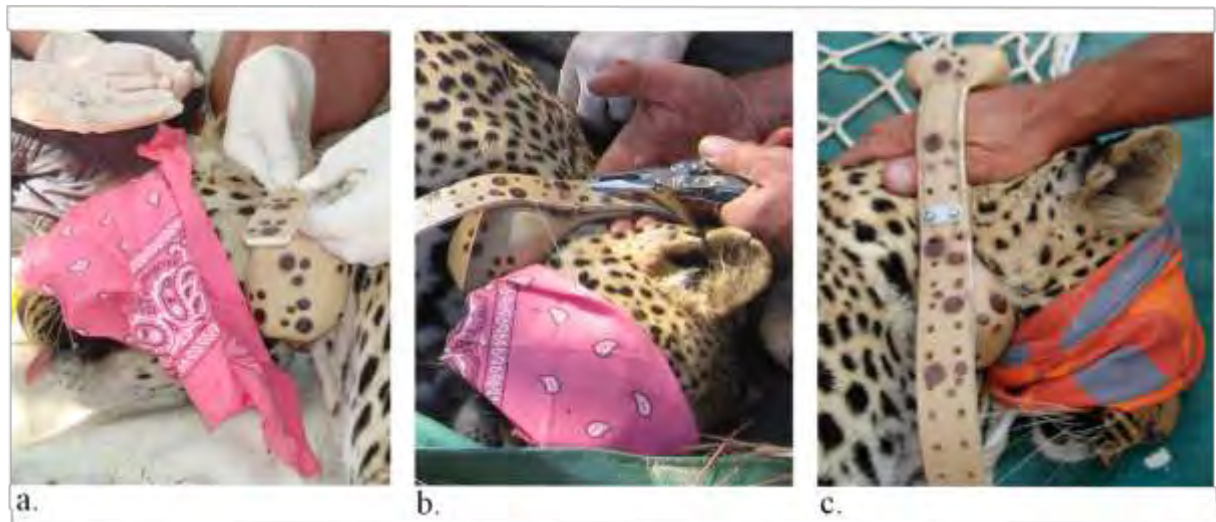
Figure 4.5: An indication of the space left underneath the collar when attached to each of the three leopards. **a.** M1, a space of three fingers was used; the collar remained on M1 until he was shot. **b.** F1's collar, a gap of about four fingers was used; this proved to be too loose as F1 dislodged the collar in a rock crevasse. **c.** F2, a large hand space was left under this collar due to F2's sub-adult status, allowing for room for her to grow into it. This was also later dislodged.

All leopards were tracked with the VHF telemetry the day after capture and all three had moved away from the site of capture. The three leopards' tracking periods overlapped for just under three weeks from 25 August to 13 September (Table 4.4).

Table 4.4: Capture dates and duration of the three collared leopards in the Mangwe area. The three leopards' tracking dates overlapped from 5 August to 13 September.

Dates	19/07 - 25/07	26/07 - 01/08	02/08 - 08/08	09/08 - 15/08	16/08 - 22/08	23/08 - 29/08	30/08 - 05/09	06/09 - 12/09	13/09 - 19/09	20/09 - 26/09	27/09 - 03/10	4/10 - 10/10	11/10 - 17/10	18/10 - 24/10	25/10 - 31/10	01/11 - 07/11	08/11 - 14/11	15/11 - 21/11
Week	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
M1	19 Jul								13 Sep									
F1			05 Aug															20 Nov
F2						25 Aug												10 Nov

4.4.2 Collar performance

M1's collar registered 319 successful fixes out of a possible 333; F1's collar registered 541 fixes out of 638, and F2's collar recorded 416 fixes out of an attempted 455 (Table 4.5). Positional Dilution of Precision (PDOP) values of five or less were recorded for 87% of M1's fixes, 84% of F2's fixes, and, due to a technical error with F1's collar, PDOP was only recorded for the first 101 fixes, of which 91% were values of five or less (Table 4.5). The first 24 hours of GPS fixes after the leopards' captures were excluded from any movement analyses, but included in home range estimates.

Table 4.5: The number of successful fixes recorded, the total number of potential fixes, the percentage of successful fixes and the percentage of good (≤ 5) Positional Dilution of Precision (PDOP) values as a measure of GPS fix precision.

Leopard ID	No. successful fixes	No. of attempted fixes	% of successful fixes	PDOP fixes of values <5
M1	319	333	96%	87%
F1	541	638	85%	*91%
F2	416	455	91%	84%

*PDOP was only recorded for F1's first 101 points due a technical error with the collar.

The number of consecutive failed fixes was investigated. F1 had the most number of failed fixes, with a maximum of nine failed attempts in a row (Table 4.6). The sets of nine, seven, six and one set of five failed attempts all occurred over a period of five days, between the 31 October and 4 November. The last two sets of five missed fixes occurred, on the 12 and 17 of November. The

highest incidence of missed fixes for all three collared leopards were single fixes (Table 4.6), of which the majority (93%) occurred during the day, between 10h00 and 18h00 (Figure 4.6).

Table 4.6: A count of the number of consecutive GPS collar fixes missed, with 1 indicating just a single fix missed, 2 indicating two fixes missed in a row, etc.

	No. of consecutive fixes missed							
Leopard ID	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**9**
M1	12	1						
F1	19	8	3	4	3	1	1	1
F2	24	3	3					
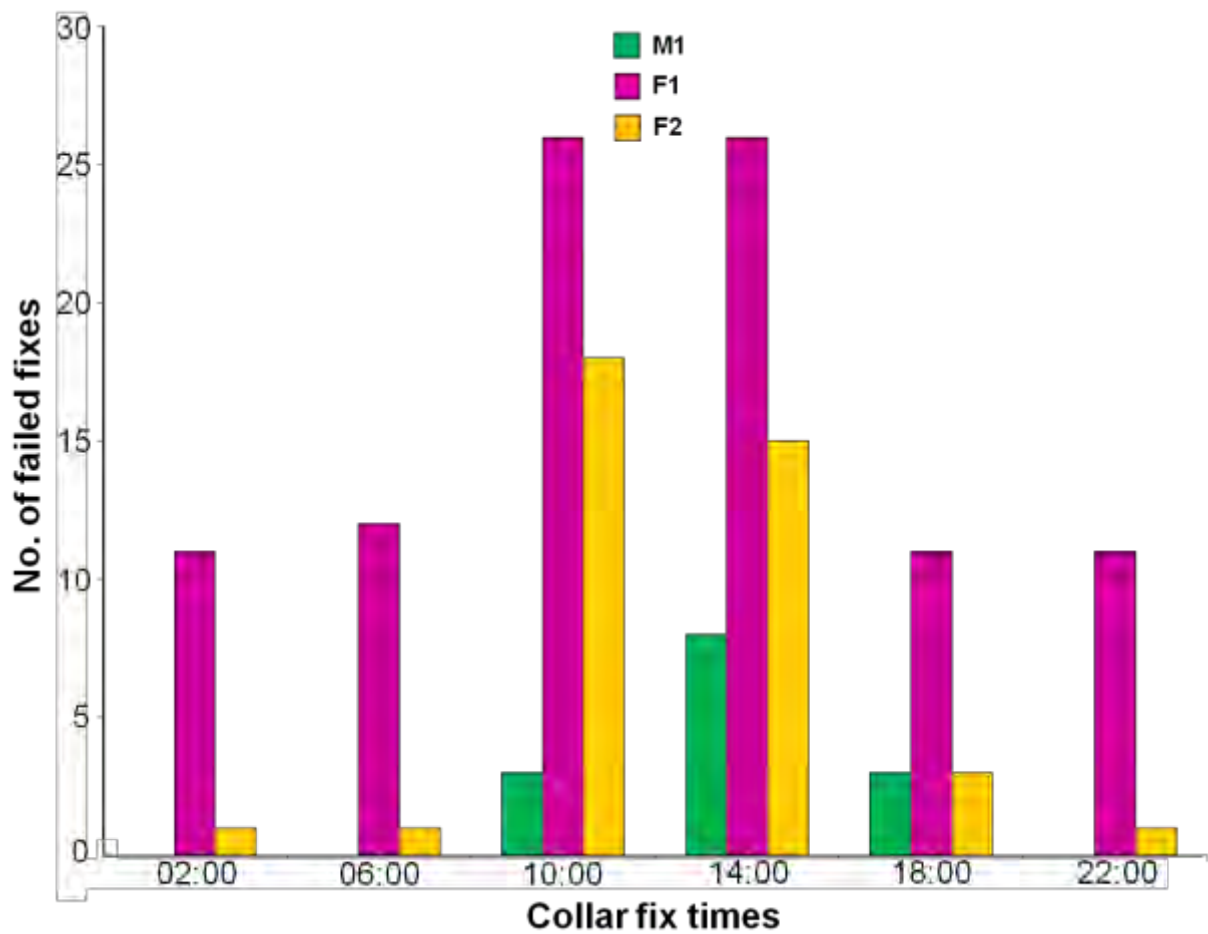
Figure 4.6: The total number of failed GPS fixes at each of the scheduled collar-fix times for each of the three collared leopards.

4.4.3 Home range estimates

Minimum Convex Polygons (MCPs)

Home range sizes calculated by the 100% MCP were 231 km^2 for the male and 32 km^2 and 37 km^2 for F1 and F2, respectively (Figure 4.7, Table 4.7). The 95% MCP home range estimates were 176 km^2 for M1, 22 km^2 and 24 km^2 for the females F1 and F2 (Figure 4.7, Table 4.7).

Table 4.7: 95% and 100% MCP home range estimates, 50% (core area) and 95% Kernel UD home range estimates and the H$_{ref}$ values used in the Kernel UD analyses for the three collared leopards.

Leopard ID	MCP (km^2)		Kernel UD (km^2)		Kernel UD H$_{ref}$ (m)
	95%	100%	50%	95%	
M1	176	231	72	263	1478
F1	22	32	11	31	495
F2	24	37	11	45	673
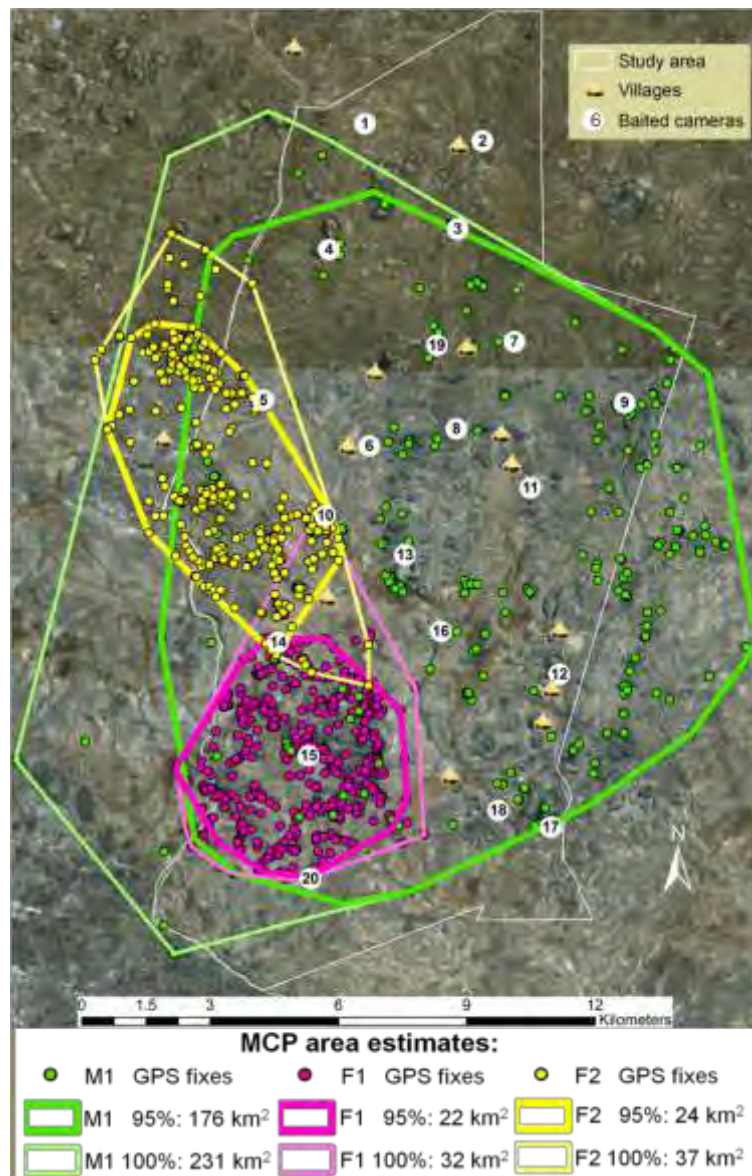
Figure 4.7: 95% and 100% MCPs and the GPS fixes for all three collared leopards. Camera-trap sites, which also correspond with cage trapping sites, and villages are illustrated.

Incremental analysis calculated that asymptotes were reached at 170 and 290 points for the male's MCP home range area. An asymptote for F1 was reached at 190 GPS fixes, and 230 fixes were needed for F2's home range size to reach an asymptote (Figure 4.8).
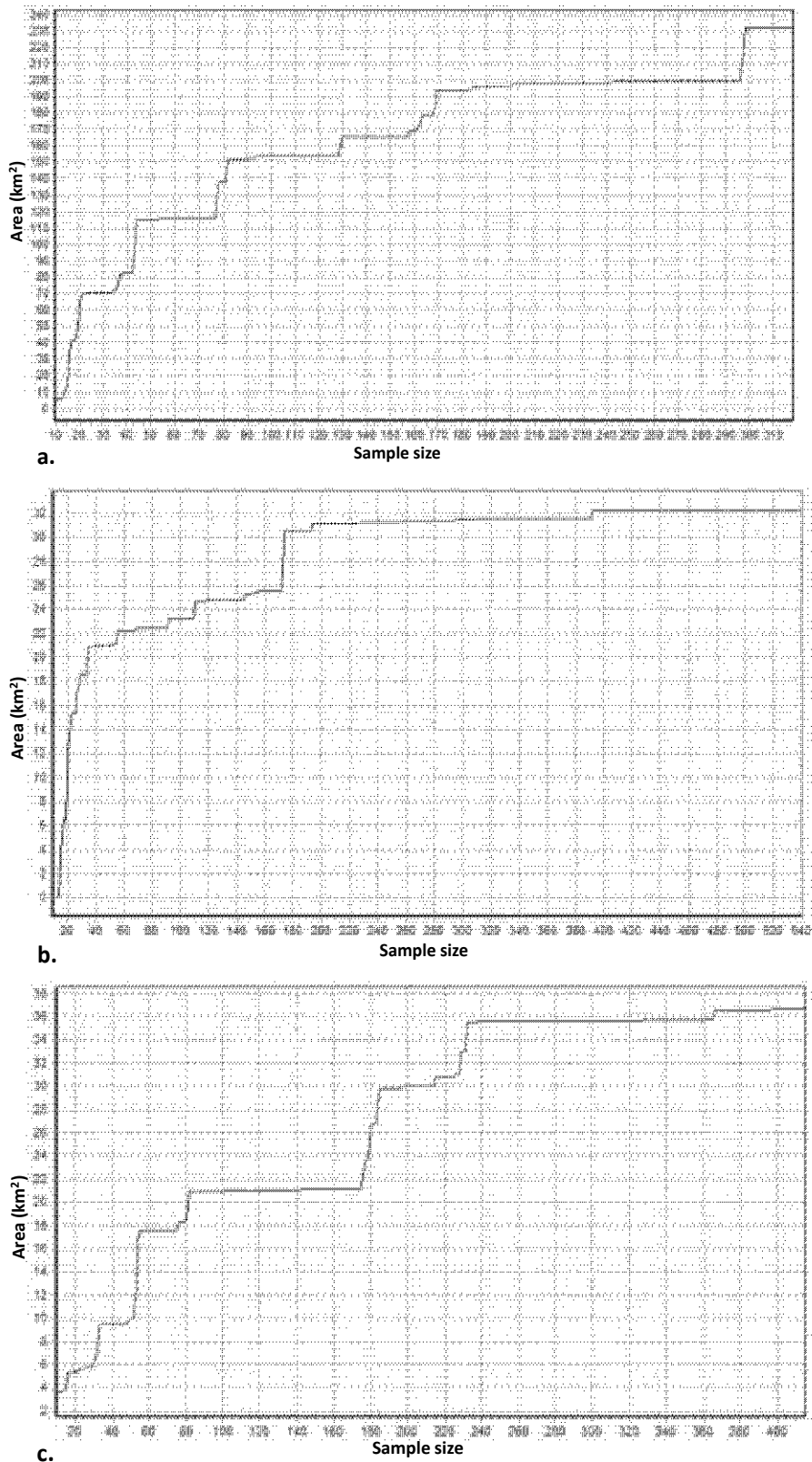
Figure 4.8: MCP incremental analysis shows that M1 (a.) reaches what appears to be an asymptote at about 170 points and again at 298; F1 (b.) reaches an asymptote at 186 points, and F2 (c.) reaches an asymptote at 225 points.

Kernel Utilisation Distributions (Kernel UDs)

Kernel UD home range estimates resulted in a 50% UD (core area) of 72 km^2 and a 95% UD area of 263 km^2 for the male, a 50% UD area of 11 km^2 and 95% UD of 31 km^2 for F1 and a 50% UD area of 11 km^2 and 95% UD area of 45 km^2 for F2 (Figure 4.9, Table 4.7).
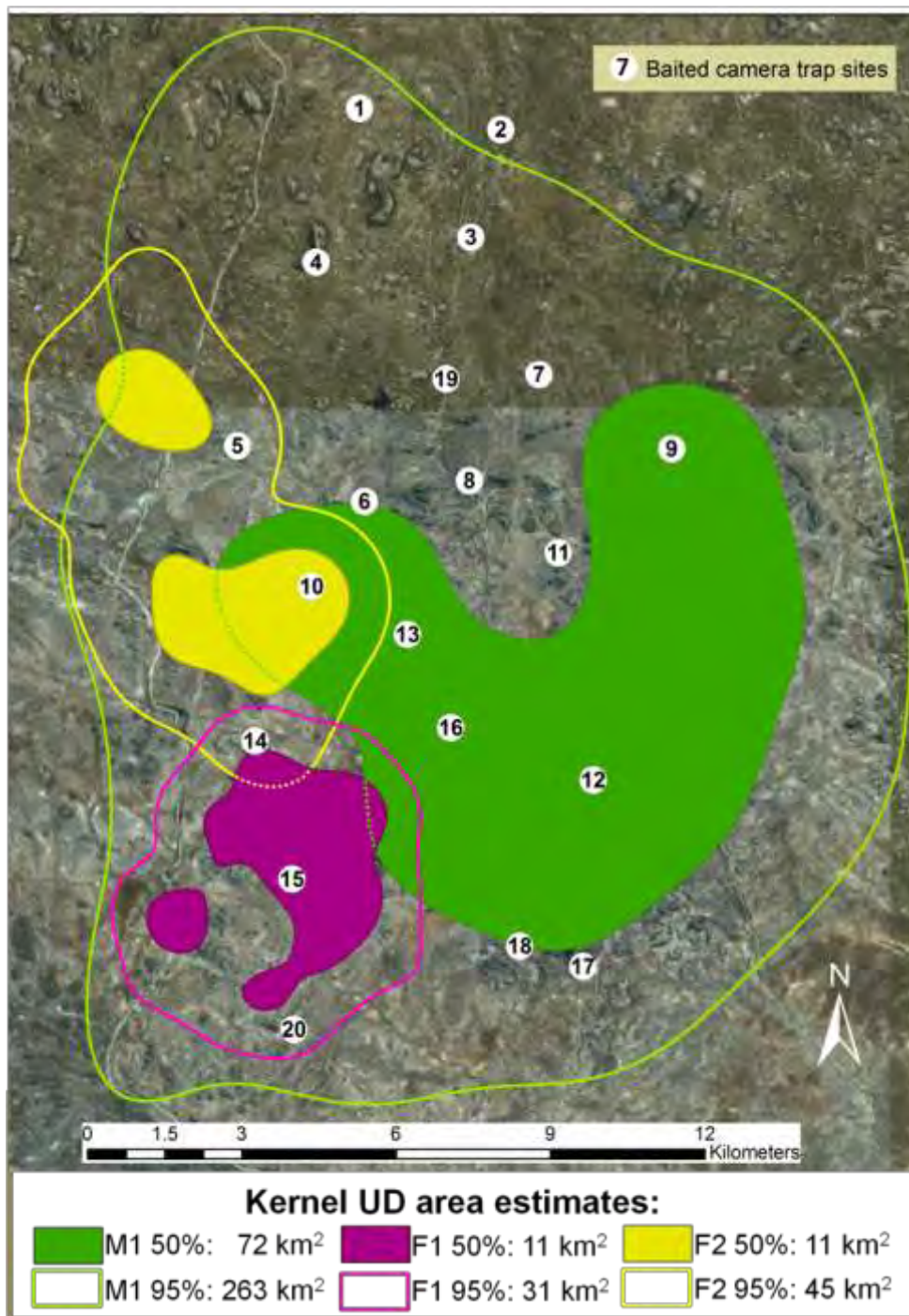
Figure 4.9: Kernel Utilisation Distribution (95% and 50%) area estimates for the three collared leopards.

Overlap among home ranges between MCPs and Kernel UDs

F1 and the male's core areas (50% Kernel UD) overlapped by 3% of the female's core range and 0.4% of the male's core range (Figure 4.9 and Table 4.8). F1's entire 95% Kernel UD area fell within the male's 95% UD (Figure 4.9 and Table 4.8). F2 and M1's core areas overlapped by 41% of the female's core range and 6% of the male's. Of F2's 95% UD range, 86% fell within the male's 95% UD range (Figure 4.9 and Table 4.8). The females' core areas did not overlap at all; their 95% UD areas did overlap each other, by 9% of F1's range and 6% of F2's range (Figure 4.9 and Table 4.8).

Table 4.8: The percentage overlap of each collared leopard's Kernel UD home range. This % is a proportion of the corresponding 95%, 90% and 50% Kernel UDs for each leopard.

Leopard ID	95% Kernel UD			50% Kernel UD		
	F1	F2	M1	F1	F2	M1
F1	-	9%	100%	-	0%	3%
F2	6%	-	86%	0%	-	41%
M1	-	-	-	0.4%	6%	-

The females' 95% MCPs overlapped the male's 95% MCP by 91% (F1) and 78% (F2) of their ranges (Figure 4.7 and Table 4.9). The females' 95% MCP overlapped each other by 5% of each of their ranges (Figure 4.7 and Table 4.9). Of the 100% MCPs, the majority of all four females' ranges (including the camera data for F3 and F4) fell within M1's range, with F1, F3 and F4's ranges all falling completely within M1's range, and 97% of F2's range overlapping M1's range (Figure 4.10 and Table 4.9). The collared females' ranges overlapped each other by 20% of F1's 100% MCP home range and 17% of F2's range (Table 4.9). The collared females' and the camera-trapped females' ranges did not overlap, based on the available data (Figure 4.10 and Table 4.9). The camera-trapped females' ranges overlapped each other by 53% of F3's 100% MCP and 15% of F4's MCP (Figure 4.10 and Table 4.9). Figure 4.10 shows that no other adult male was detected by the camera-traps within the collared male's core area during the study.

Table 4.9: The percentage overlap among the three collared leopards and the two camera-trapped females. This table presents the percentage of overlap between the corresponding home range estimates, where available, as a proportion of each leopard's estimated home range.

Leopard ID	Overlap between 100% MCPs					Overlap between 95% MCPs			Overlap of Core Areas with MCPs				
	F1	F2	F3	F4	M1	F1	F2	M1	F1	F2	F3	F4	M1
F1	-	20%	0%	0%	100%	-	5%	91%	-	16%	-	-	-
F2	17%	-	0%	0%	97%	5%	-	78%	5%	-	-	-	-
F3	0%	0%	-	53%	100%	-	-	-	-	-	-	-	-
F4	0%	0%	15%	-	100%	-	-	-	-	-	-	-	-
M1	16%	14%	4%	14%	-	-	-	-	8%	9%	6%	29%	-
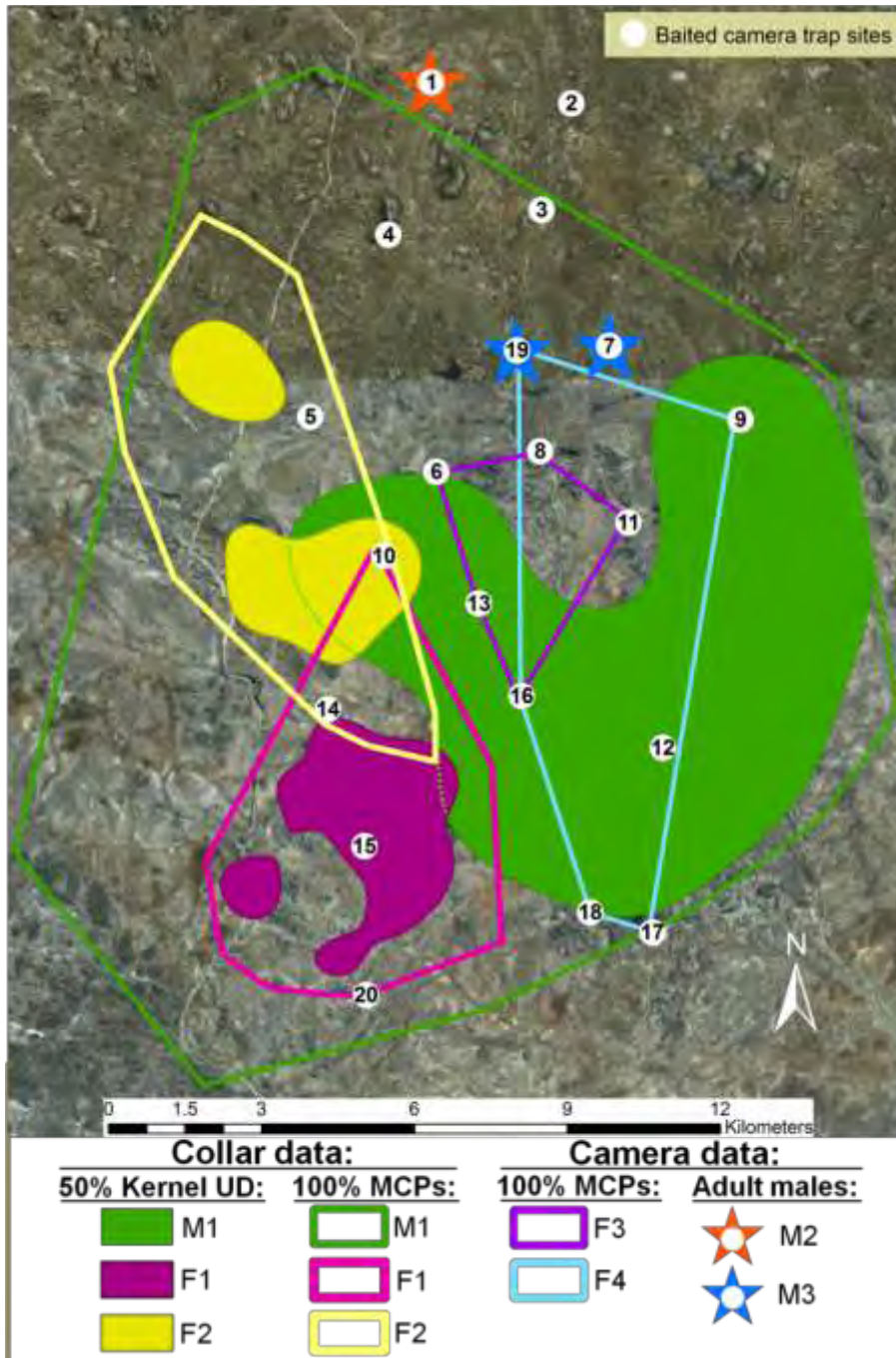
Figure 4.10: Overlap in the home range estimates of three collared and two camera-trapped leopards. Data for the three collared leopards shows their Kernel UD core home ranges (50%) and their 100% MCPs. Camera-trap data provided a coarse 100% MCP for two females captured at four (F3) and five (F4) camera sites. The stars indicate the location of the two other mature males 'captured' in the area, based on camera-trap data (other than collared M1).

4.4.4 Movement patterns

Daily movements differed significantly among individuals ($p < 0.001$, $H_{2, 220} = 20.35$). The male moved significantly further than the females (F1, $p < 0.001$; F2, $p = 0.004$) (Figure 4.11). Distances moved by the two females did not differ significantly ($p = 0.696$) (Figure 4.11).
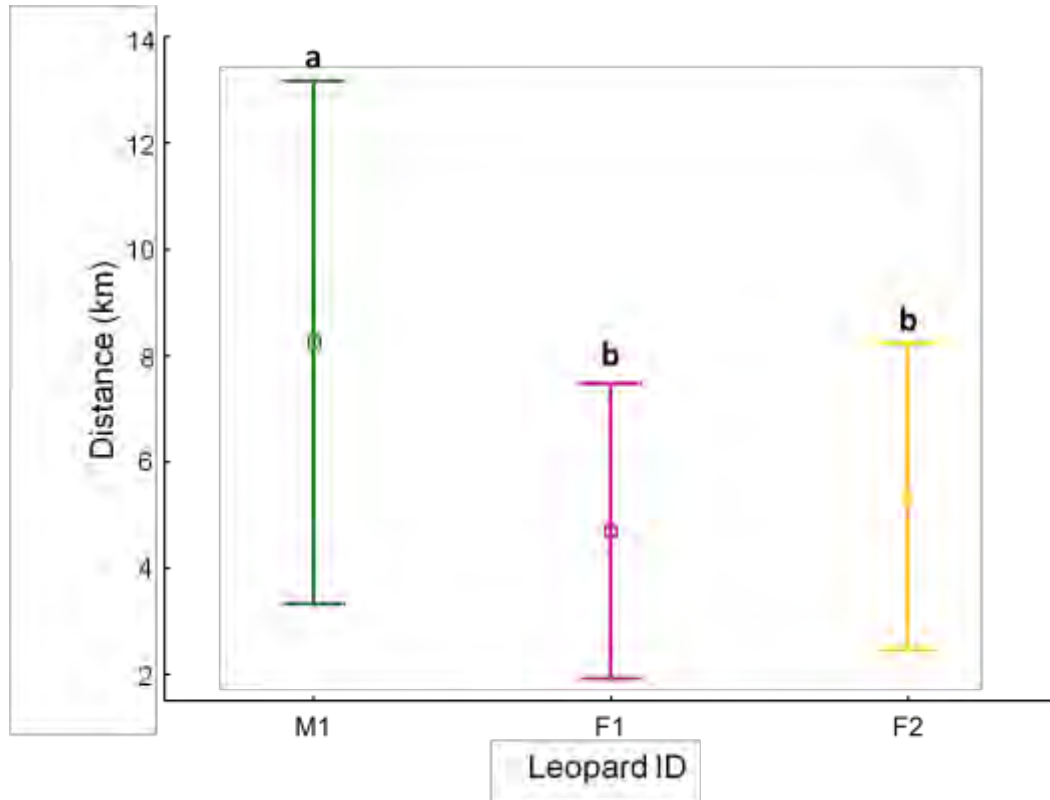
Figure 4.11: Daily distances moved by the three leopards (km) were significantly different between the male and both female leopards, but not significant between the two females. Letters (a and b) above the data indicate significantly different individuals. Data points are means, and lines ±SD.

The mean daily distance moved for the male was 8.3±4.9 km with a maximum distance moved of 19.7 km. M1 travelled a total measured distance of 45.5 km over a period of 55 days (Table 4.10). The mean daily distances travelled for F1 and F2 were 4.7±2.8 km and 5.4±2.9 km, respectively, with maximum distances per day of 11.7 km and 11.8 km. F1 travelled a total of 43.3 km over a tracking period of 92 days, and F2 travelled a total distance of 39.2 km over 73 days (Table 4.10).

Table 4.10: The mean, minimum and maximum distances moved between GPS fixes per day for the three leopards. The total distance travelled during the tracking period and the number of tracking days analysed for each leopard are also shown.

Leopard ID	Distance moved between GPS fixes (km)			Total distance travelled (km)	Tracking period (No. of days)
	Mean	Minimum	Maximum		
M1	8.3±4.9	0.07	19.7	45.4	55
F1	4.7±2.8	0.05	11.7	43.3	92
F2	5.4±2.9	0.2	11.8	39.2	73

The null hypothesis that there were no significant differences among the distances moved at different times of the day, was rejected based on the Friedman's ANOVA ($p < 0.001$); indicating that the distances moved by each leopard were significantly different across the different times of the day (Figure 4.12). For all three leopards, distances moved at night were greater than the distances travelled during the day (Figure 4.12). The shortest distance travelled was measured between fixes during the day, namely between 10h00 to 14h00, and 14h00 to 18h00 (Figure 4.12).
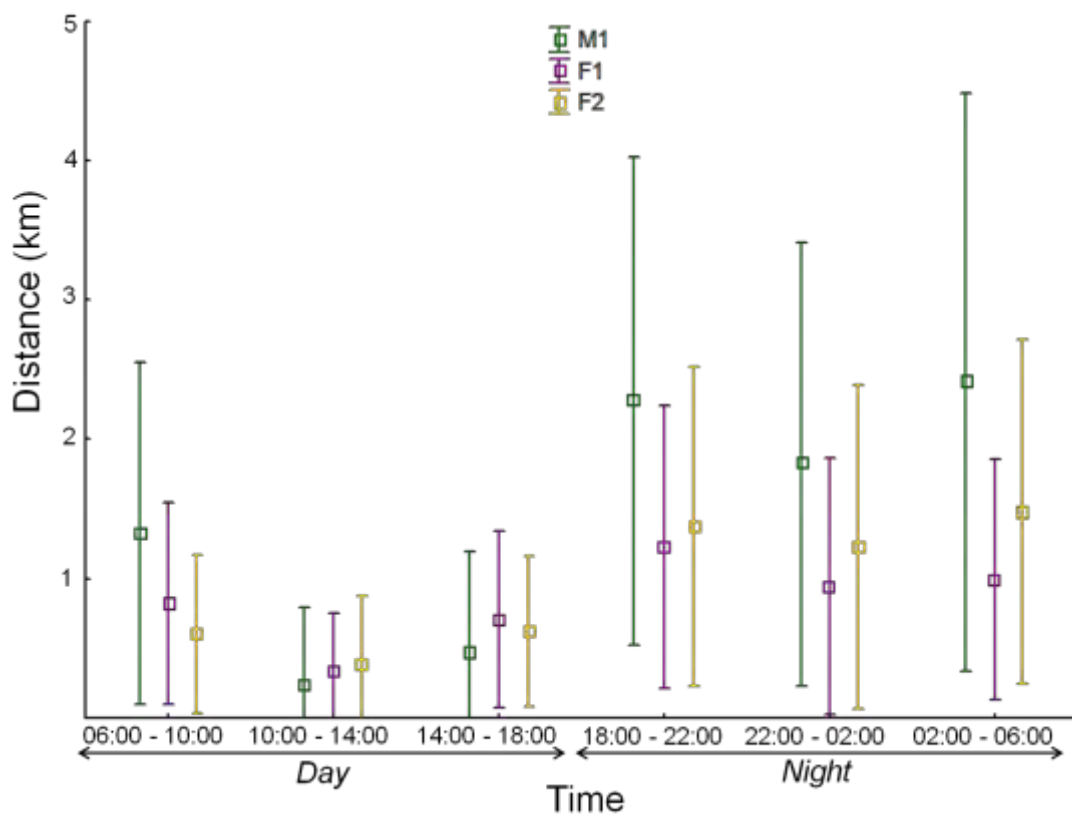
Figure 4.12: Distance travelled by the three leopards (km) between fixes, showing the diel movement patterns. Data points are means, and lines ±SD.

4.5 Discussion

4.5.1 Capture and immobilisation

Cage-trapping proved to be a successful method of live capture in order to attach GPS collars to the leopards in this study. However, the method required significant effort. The initial cage-trapping effort of 225 trap nights, plus the extra 100 trap nights spent trying to capture an additional male after M1 was shot, represents an intensive trapping effort of 108 trap nights per leopard. This is much higher than the effort documented in other studies, such as one leopard capture per 2.4 trap nights for Steyn (2007), 5.2 trap nights for Hamilton (1976) and 21.5 trap nights for Bailey (1993). Even the initial effort (before the male was shot) of 75 trap nights per leopard capture is much higher than these studies. However, it is considerably less than the 341.3 trap nights per leopard spent by Grimbeek (1992) and 833.3 of Norton and Henley (1987). The high capture rate of the first three studies mentioned is most likely due to a combination of factors which influence the leopard densities and behaviours in those areas. All three areas are conservation areas where leopards are not actively persecuted; two of the studies were based in national parks (Hamilton 1976, Bailey 1993), and Steyn's (2007) study was based in a private eco-tourism reserve. Therefore, the degree of protection afforded to leopards in these areas, and the relatively mesic habitats, may have resulted in higher leopard densities (5 − 17 leopards/ 100 km^2), thereby increasing the chances of successful captures (Hamilton 1976, Bailey 1993, Steyn 2007). Carnivores that are not exposed to regular persecution may also be less wary than those regularly targeted by hunting, poisoning and/or snaring practices. They may be less suspicious of vehicles, human scents or the metal of the cage traps, resulting in less effort required to capture them (Frank *et al.* 2003).

Steyn (2007) credits his successful capture rate to the pre-baiting method employed and the high leopard density in the Northern Tuli Game Reserve (NTGR). The low capture rates of Grimbeek's study (1992) were attributed to the wariness of the leopards in the area due to their continuous exposure to persecution, large home ranges, non-target catch species which hindered the capture process and the high abundance of natural prey in the area. Although Norton & Henley (1987) estimate relatively high densities for the Cedarberg Wilderness Area (6-9 leopards/100 km^2), their capture effort was very high. If these density estimates are accurate, then it is likely that the low capture rate was due to trap-shyness of the leopards as a result of their extensive persecution on the neighbouring livestock farms (Norton & Henley 1987).

Trap-shyness was highly likely in my study as leopards have been heavily hunted in the area for the last 20 years (Grant 2008). Leopards have also been killed as problem animals by the livestock farmers, and poached, using methods such as gin-trapping, cage traps, wire snares,

shotgun traps and poison (Grant 2008). This persecution is very likely to have increased the wariness of leopards, in this area, towards foreign objects and smells, especially the older animals. All three leopards were caught in the dry winter months, with no success when attempting captures in the summer. This could be attributed to the fact that during the wet summer months, ungulate calves were more abundant, providing relatively accessible natural prey for the leopards, and thereby decreasing their need to scavenge from baits in the cage traps (Grant 2008, Swanepoel 2008).

Cage traps were the chosen method of live capture in this study as they are simple to set up and did not require any specialised skills or training. However, injury rates in cages are often high, as leopards can seriously damage their claws and canines in an attempt to escape (Frank *et al.* 2003). None of the leopards endured any extensive damage to their canines, and none of their claws were completely broken off. F1 suffered the most damage to her claws by splitting five of them; but this damage appeared to be minor and was unlikely to impede her natural hunting success. The canine and claw damage to the animals could be reduced by limiting the amount of time that the animals are constrained. An alarm system such as the one employed by Steyn (2007) could be used to alert the team as to when the trap door is set off. This way, the capture team can respond immediately, thus reducing the time of containment in the cage for the animal. The blankets used in the approach and covering of the cage were very effective in this study.

Free-darting, an alternative capture method, has been successfully employed in other studies (Hunter *et al.* 2003, Balme 2009). Free-darting of carnivores off a bait is usually attempted from the safety of a vehicle, to which a target animal will need to be habituated, otherwise it will flee before darting (Hunter *et al.* 2003, Steyn 2007). Free-darting of leopards off a bait from a 'hide' was considered. However, this was dismissed as being too risky as it would have had to have been attempted at night, and the risks involved in searching for a partially sedated leopard were considered too high. Another capture method that is being used more frequently in carnivore live captures, and with great success, is the use of soft-hold foot snares (Frank *et al.* 2003). This method involves a steel cable snare which is anchored by steel rods either to a tree or in the ground (Frank *et al.* 2003). The advantages of this method are that injuries to the target animal are minimized, the foot snare equipment is easily transportable and relatively inexpensive, and it can be easily camouflaged in order to target wary carnivores (Frank *et al.* 2003, Balme 2009). However, the correct set-up of this method is necessary for successful captures and the safety of the target carnivores as well as the personnel involved. This requires a certain level of skill and experience (Frank *et al.* 2003). Another disadvantage is that any potential dangerous carnivore captured, would have to be immobilised before release (Frank *et al.* 2003), and this can be both

time-consuming and costly. The equipment and skilled personnel required for foot snaring were not available for this study.

The weights of the captured leopards were similar to those estimated by most other southern African studies. However, the male (M1) was slightly heavier than the average male weight (Table 4.11). M1 was heavier than male leopards from Tsavo National Park; however, the females from Tsavo were heavier than F1 and F2 (Hamilton 1976). Both the male and females in Steyn's (2007) study were slightly heavier than the male and females in my study. The smallest leopards, in terms of mass, were measured by Stuart (1981) in the Cape Province of South Africa. However, these leopards are renowned for their petite size in comparison to the rest of Africa's leopards. Marker and Dickman (2005a) demonstrated that leopards in Namiba were also smaller than those found in other areas. The average male weight of the leopards in Wilson's study (1976) is also comparatively low. However, the females fall within the average range of the other studies.

Table 4.11: Weights of leopards from this study and other African studies.

Author	Study Area	Average weight (kg) and number of leopards sampled (*N*):		
		Adult male	**Adult female**	**Sub-adult female**
Grant 2012 (This study)	Mangwe District, Zimbabwe	62 (1)	35.5 (1)	29 (1)
Bailey 2003	Kruger National Park, Mpumalanga, South Africa	58.2 (3)	37.2 (6)	30.1 (3)
Hamilton 1976	Tsavo National Park, Kenya	55.6 (7)	40 (3)	
Hunter *et al*. 2003	Phinda Private Game Reserve, Kwa-Zulu Natal, South Africa	58.5 (4)	31.5 (5)	
Marker & Dickman 2005a	North-central Namibia	45.9 (6)	30.3 (5)	
Steyn 2007	Northern Tuli Game Reserve, Botswana	63.6 (1)	37.2 (6)	
Stuart 1981	Cape Province, South Africa	30.9 (30)	21.2 (6)	
Wilson 1976	Zambia	49 (9)	33.6 (6)	

4.5.2 Collar performance

The GPS collars performed satisfactorily. However, the premature failure of F1's UHF receiver and F2's VHF transmitter were reasons for concern. F1's collar was unable to communicate with the download console, even once I had retrieved the collar. Had F1 not shed her collar I would not have been able to retrieve the data until she was recaptured. F2's collar could not be retrieved after she shed it. Based on previous tracking experiences, the VHF signal could usually be used

to detect the collars at a distance of at least 12 km from an elevated position, and therefore should have been able to pick up F2's collar if she had shed it anywhere within her MCP range.

Of the total number of fixes, the failure rate ranged from 3% to 15%. This is less than the failure rates recorded in other studies which used GPS collars on leopards. Failures of between 15 and 29% were recorded in the Waterberg Mountains (Swanepoel 2008) and 46 to 59% in the Soutpansberg mountain range (Chase-Grey 2011). Both of these studies attribute their failure rates to the mountainous terrain which dominates the areas, and probably caused some interference with the GPS collar's direct line to the satellites (Swanepoel 2008, Chase-Grey 2011). It is likely that the terrain, as well as the leopards' behaviour, influenced the failure fix rate in the present study. Of the single failed fixes (when the collars were only temporarily blocked from the satellites), 93% occurred during the daylight hours. This is most likely due to the leopards resting during the daytime, when they are either prostrate, which could inhibit the collars' ability to communicate with the satellites, or when they were resting in the caves or crevasses of the kopjes (Swanepoel *et al*. 2010). F1's collar had the most successive failures, up to nine missed fixes; and this prolonged obstruction of the collar could be explained by the suspected denning behaviour towards the end of her tracking period. Female leopards' den sites are often in areas of very dense vegetation, caves or crevasses (Bailey 1993, Skinner & Chimimba 2005) and this is a likely explanation for the numerous successive failed fixes which occurred over the five-day period when she was tracked, via VHF, to the same location near the summit of a kopje.

4.5.3 Home range estimates

The overall home range estimates follow a similar pattern to the one observed in previous studies, where males have larger home ranges than the females (Hamilton 1976, Bailey 1993, Bothma *et al*. 1997, Mizutani & Jewell 1998). The combined collar and camera-trapping data support the hypothesis that one male will generally have three to four overlapping, resident females within his range (Bothma & Walker 1999). The 95% Kernel UDs were larger than the 100% MCPs for M1 and F2, and only slightly smaller than F1's 100% MCP.

The home range estimates in the current study are comparable to those of leopards studied by Mizutani and Jewell (1998) in the scattered woodland and grassland of the Lolldaiga Hills ranch in Kenya. However, their male home ranges were smaller than M1's (64 to 110 km^2). This could be attributed to the unusually high density of leopards in the area (12.5 leopards/100 km^2, including cubs), where the males ranges were probably delimited by their neighbours (Mizutani & Jewell 1988). Therefore, although the habitat and landuse of the Lolldaiga Hills ranch are

similar to that of my study area (the Lolldaiga area is characterised by a semi-arid ecological zone, with scattered woodland dominated by *Vichellia* species and a pastoralist/village based landuse), the prey biomass and variety appear to be much higher than those of my study area (Mizutani & Jewell 1998).

The home ranges in the current study are smaller than those estimated in arid regions, such as the Kalahari Gembsbok National Park (Bothma *et al.* 1997) and the Waterberg Plateau in Namibia (Marker & Dickman 2005a). Interestingly, the home ranges of leopards in the mountainous region of the Waterberg in Limpopo (Grimbeek 1992, Swanepoel 2008) have comparatively large home ranges, especially the females, as do the leopards tracked in the Stellenbosch region of South Africa (Norton & Lawson 1985). This suggests that although an area may fall within a more mesic landscape, the landuse in that area (e.g. agriculture) can affect prey availability, suitable habitat and therefore, leopard behaviour. It also suggests that the Mangwe area provides more suitable and higher-quality habitat than other more arid regions (Bothma *et al*. 1997, Marker & Dickman 2005a). This could be the result of the high density of kopjes which provide a suitable habitat for leopards and their prey. An unpublished and ongoing leopard home range study on a game sanctuary in the Mangwe area (Peek & Grant unpubl. data) estimated a 95% Kernel UD of 50 km^2 and a 50% Kernel UD of 13 km^2 for an adult female, based on eight months of radio-tracking data. This home range estimate is very similar to my study, which is 40 km to the south. This comparison is useful, as it substantiates the limited data produced from the area, and enables relatively reliable extrapolations to made for the Mangwe area, delineated by the former Marula ICA (Chapter 5: Figure 5.1).

The home ranges in the current study were larger than those reported for the mesic habitats of the Kruger National Park (Bailey 1993), the forests of Nepal and Thailand (Odden & Wegge 2005, Simchareon *et al.* 2008) and the Matopos National Park to the east (Smith 1977). These areas are all characterised by high quality habitats, with high prey biomass and are formally protected as national parks. This has resulted in a higher density of leopards, whose metabolic needs are met within comparatively small areas.

It is generally acknowledged that the MCP is an outdated and less accurate home range estimation method, and should not be used anymore for home range estimates (Börger *et al.* 2006). It is heavily influenced by variations in sampling regimes, and Börger *et al.* (2006) found that its variance changes unpredictably from between -10% to +400% due to increased sampling effort. However, the results of my MCP estimates and Kernel UD estimates were relatively similar, which lends some confidence to the use of both of these estimates in the current study.

The Kernel UD results in more accurate and precise estimates (Börger *et al.* 2006, de Solla *et al.* 1999) as well as providing information on how the animals utilise these home ranges by analysing the time spent in specific areas (Seaman *et al.* 1999). This provides a better understanding of the ecological resources that are valuable to the leopards in terms of habitat selection, or it could identify areas that leopards avoid, all of which would influence spatial patterns (Jackson 1996, Seaman *et al.* 1999). Habitat use was not investigated in the current study due to the lack of current aerial photographs from which habitats could be mapped. However, based on the visual interpretation of the home ranges and GPS fixes layered onto topographic maps and Google Earth images, it appears that the leopards favoured kopje and dry riverbed habitats. They also appeared to show some avoidance of the cattle out-posts/villages where humans and domestic dogs could have been a deterrent. Further research into habitat preference and avoidance, and the influence of human settlements on movement, would be an important aspect of leopard ecology to investigate in the Mangwe area.

Another issue, when comparing home ranges, is that most of the variation in these estimates is due to the differences between each individual's behaviour and each study area's landscape and prey (Börger *et al.* 2006). Therefore, perhaps more effort should be directed at tracking more individuals within the study, as opposed to tracking fewer individuals for longer periods of time (Börger *et al.* 2006). Also, caution must be taken when comparing these individual's ranges and study areas, as these may not be representative of the general population or habitat, but rather specific to the individuals of one particular area (Börger *et al.* 2006).

Unfortunately, seasonal changes in home ranges could not be investigated due to the relatively short tracking periods. It would be valuable for future studies to investigate seasonal changes in leopard spatial utilisation in the area, and how this may impact upon the human-carnivore conflict in the area.

Home range overlap

My overlap data are comparable with other studies which have demonstrated that there there is considerable overlap between females and males, yet little to no overlap amongst males (Bailey 1993, Marker & Dickman 2005a). This is a common pattern in other solitary felid studies, where resident males maintain exclusive core ranges, showing little overlap with neighbouring males (Bailey 1993, Jackson 1996, Mizutani & Jewell 1998, Silwa 2004, Dillon & Kelly 2008). The degree of overlap is important when determining possible densities of mature males in an area for off-take purposes, and this is discussed in Chapter 5.

The degree of home range overlap can be influenced by a number of factors. In order for felids to maintain exclusive ranges, food and water resources must be evenly distributed and stable throughout the year (Mizutani & Jewell 1997). This is rarely the case, and usually habitats are heterogeneous, with some areas rich in resources and others, poor. Therefore, some degree of overlap is to be expected as leopards access the richer areas (Jackson 1996). However, it has been reported that in more mesic areas, where resources are abundant, there is less need for home ranges to overlap extensively (Bailey 1993). By contrast, in more arid areas, where resources are limited, leopard home ranges may overlap more extensively as individuals vie for the limited resources (Stander *et al*. 1997). High degrees of overlap were recorded for leopards in Namibia by Stander *et al*. (1997). However, it appears that any social encounters between males were avoided, as they concentrated their activities in the overlapping areas at different times. These data support the trend that dominant male leopards maintain exclusive ranges, with the main aim of maintaining exclusive breeding rights to the females within these ranges (Bailey 1993, Stander *et al*. 1997, Mizutani & Jewell 1997).

Another factor reported to affect the degree of overlap among leopards is persecution (Marker & Dickman 2005b). This has been reported to result in range expansion and an increase in the overlap of territories of immigrants (Tuyttens *et al*. 2000, Marker & Dickman 2005b). With leopards, the removal of a dominant male creates a vacant territory that subordinate sub-adults or immigrants can colonise, resulting in a higher degree of overlap due to their yet-to-be-established dominance (Bailey 1993, Athreya 2006). This also impacts on cub survival rates, as the new males entering the area will often kill any previously sired cubs (Loveridge *et al*. 2007).

4.5.4 Movement patterns

Mean daily distances travelled by the collared leopards in the present study are comparable (albeit slightly lower) with Grimbeek's (1992) study from the Waterberg mountain area of Limpopo of 8.6 km (male) and 6.1 km (female). Mean daily displacement distances from the current study are also less than those reported by Stander *et al*. (1997) in Namibia, where males travelled a mean daily distance of 12.2 km and females 6.8 km, and Bothma and le Riche (1984) from the Kalahri of 14.3 km (males) and 13.4 km (females). The results from my study are much larger than those reported by Bailey (1993) of 2.8 km (males) and 1.5 km (females) and Hamilton (1976) of 4.2 km (males) and 2.3 km (females).

It is evident that the studies that demonstrated larger home ranges, from the more arid areas, have higher mean daily distances travelled (Bothma & le Riche 1984, Stander 1997). Similarly, those with smaller reported home ranges sizes, have smaller daily displacements (Hamilton

1976). The primary motivation for leopards to travel would be to satisfy their prey requirements, and, once again, prey abundance essentially structures a leopard's spatial ecology (Bailey 1993). Males will also traverse their ranges in order to assess the reproductive state of the females within his range (Bailey 1993, Mizutani & Jewell 1998), and therefore in areas with high leopard densities, males will travel shorter distances to access these females. Hence, in areas with higher prey biomass and leopard densities, like the Kruger National Park, South Africa (Bailey 1993), leopards would have smaller daily displacement distances than leopards in more arid areas (e.g. the Kalahari, Bothma & le Riche 1984) where prey densities and leopard desnities would be significantly lower, thus forcing leopards to travel further to encounter prey (Bailey 1993, Swanepoel 2008).

The maximum distance a leopard would travel would also be influenced by other leopards in the area, where encounters with other leopards are generally avoided, thus confining travels to within each leopard's territory (Bailey 1993). Males' ranges are usually larger, as they patrol these territories, scent-marking and calling, often travelling at speeds of about to 2.9 km/hour (Bailey 1993). By contrast, females generally move relatively short distances, moving the minimum distance that it takes to obtain prey and rear their young (Mizutani & Jewell 1998).

The diel patterns of leopard movement in my study could only be analysed in terms of the four-hourly intervals that the collars were set to, and the distances travelled between these times. However, there was a significant difference between day-time and night-time movements, highlighting the increased distance moved by all three leopards at night. The male and F2 moved the greatest distance between 02h00 and 06h00, and F1 moved the most between 18h00 and 22h00. Grimbeek (1992) reports that movement peaked at 18h00 for the female and 04h00 for the male, showing a similar pattern to the leopards in my study. A review of literature by Hayward & Slotow (2009) indicated that leopards have minor crepuscular peaks in activity, with three distinct activity periods between 21h00-02h00, 19h00-20h00 and 03h00-06h00. This is similar to my findings, where leopards moved greater distances at night, and at dawn and dusk, than they did during the day. Although leopards are thought to be strictly nocturnal, they are often quite active during the day, sometimes even hunting in the day (Bothma & Walker 1999). This usually happens in protected areas where leopards are not persecuted and have become habituated to people (e.g. Londolozi and Kruger National Park), and also in areas with low human densities (Balme 2009). Some movement was recorded for the leopards in my study during the day.in my study. Therefore, my data supports the notion that leopards are generally nocturnal, but may be active during the day when conditions are suitable.

4.5.5 Conclusion

The home range estimates in this study resulted in a large home range of 263 km^2 for the adult male and a range of 31 km^2 for the adult female and 45 km^2 for the sub-adult female. The results suggest that females in the Mangwe area have overlapping home ranges, with exclusive core areas. It appears that about four to six females' ranges overlapped the range of the adult male. Due to insufficient data, it was not possible to conclude that adult males' ranges did not overlap in the study area. For future studies it is recommended that more individuals and more study areas are monitored. Home range estimates should be analysed using the more robust Kernel UD method, as the repeatability and comparability of these results rely on detailed standardised sampling regimes and reporting methods.

CHAPTER 5

SYNTHESIS

AND

MANAGEMENT RECOMMENDATIONS

The view to the north of the study area as seen from atop 'Nyongeshava' (Smooth hill).

5.1 Synthesis

5.1.1 General aim and results

This study aimed to provide robust data on leopard density and spatial ecology in a mixed landuse area of Zimbabwe, in order to aid conservation managers in setting sustainable off-take quotas.

5.1.2 Spoor survey

Spoor index surveys in this study provided what appear to be underestimates for the leopard population (3.3 leopards/100 km^2). Based on the limitations of spoor index surveys, camera-trapping appears to be the more robust population density estimation method. However, the spoor index calibration equations used in my study were based on areas with different landuse patterns and leopard densities and this may have affected my results. In order for the method to be utilised successfully, spoor indices would need to be calibrated for the area against a true density estimate. If a reliable calibration equation could be calculated, I believe that the method could provide a relatively low-cost and repeatable approach for estimating and monitoring the leopard population in the Mangwe area.

5.1.3 Camera-trapping survey

Unbaited camera-trapping was not a successful leopard density estimation method in this study. The unbaited camera survey produced only six leopard photographs, with only two being suitable for identification purposes. However, the method did provide a useful mammal inventory for the area, for which no previous data existed. Previous studies have successfully captured leopard photographs using unbaited cameras (Henschel & Ray 2003, Steyn 2007, Balme *et al*. 2009a), but the extra effort required in this study would not have been feasible. By contrast, baited camera-traps were extremely successful in capturing identifiable leopard photographs, producing 292 identifiable leopard photographs. Based on the various factors affecting this method and the methods of analysis, it appears that the programme CAPTURE's M_h model using the MMDMOSA and the programme SPACECAP using a buffer of 2.5 km (HMMDM) produced the most accurate density estimates for the area (4.79±0.83 and 5.12±0.62 leopards/100 km^2, respectively).

5.1.4 Capture and immobilisation

Cage trapping was a successful method for the live capture of leopards in this study. However, much effort was required (225 trap nights) and the method proved unsuccessful during the

summer months, when natural prey was more plentiful. In order for cage trapping to be employed successfully, captured leopards should spend as little time in the cages as possible, with the main aim of the capture and processing procedure being to minimise the stress on the animal. An alarm system that alerts one as to when the cage door is triggered would be beneficial for reducing the time spent in the cage (Steyn 2007). In addition, personnel approaching the cage behind blankets and covering the cage appeared to reduce the stress of the leopards, allowing the veterinarian to approach and dart the leopard with relative ease. The possible damage that can result from the leopard trying to escape the cage appears to be one of the greatest draw-backs to this technique. For this reason, I would recommend attempting the alternate method of soft-catch snares as this method has been shown to be extremely successful, both in terms of capture effort and reducing injury to the animals (Frank *et al.* 2003, Balme 2009).

5.1.5 Home range estimates

Although it has been advised not to use MCPs to estimate animal home range sizes (Börger *et al.* 2006), they are often required in order to compare results with previous studies. MCPs in the current study performed well, resulting in similar overall home range estimates to those of the 95% Kernel UD. However, the Kernel UDs do provide a more accurate home range estimate and also provide more meaningful information such as the proportion of time an animal spends in any area of its range (Worton 1989, Seaman & Powell 1996, Swihart & Slade 1997, Seaman *et al.* 1999, Rodgers & Kie 2011). In my study area, the male leopard had a 95% Kernel of 263 km^2 and a core area (50% Kernel UD) of 72 km^2 which appeared to exclude any other adult males. The collared females had 95% Kernel UDs of 31 and 45 km^2, with core areas of 11 km^2 each. The females' core areas did not overlap with each other, but did overlap with the male's core area. When the camera-trapping data was plotted together with the home range data, it appeared that four to six females' ranges overlapped with the one collared male's home range. This pattern is similar to the one demonstrated by Bailey (1993) for leopards in the Kruger National Park in South Africa and Mizutani and Jewell (1998) for leopards in the north-west of Kenya. My data support the theory that although leopards are solitary felids, some form of social congruency exists, which is determined by the territoriality and home range priorities of the sexes (Etkin 1967, Bailey 1993, Macdonald *et al.* 2010b).

5.1.6 Movement patterns

Mean daily distances travelled were significantly greater for the male (8.3±4.9 km) than the two females, who travelled 4.7±2.8 km (F1) and 5.4±2.9 km (F2), respectively. It appears that leopards' daily displacements are proportional to their home range sizes, where the larger the

range size, the greater the required daily displacement in order to satisfy prey requirements. My estimates of home range and daily displacement are smaller than those recorded in more arid areas (Bothma & le Riche 1984, Stander 1997), but greater than those recorded for more mesic areas (Hamilton 1976, Bailey 1993). Males' ranges are generally larger, as they patrol their boundaries and locate females, whilst females generally travel the minimum distance required to obtain prey and successfully rear their young (Bailey 1993, Mizutani & Jewell 1998). For all three collared leopards, the distances travelled at night were significantly greater than those travelled during the day, and the greatest distances moved were between 02h00 and 06h00 for the male and F2, and 18h00 and 22h00 for F1. However, some day-time activity was recorded, suggesting that, although the leopards in my study appear to be mostly nocturnal, they are occasionally active during the day.

5.1.7 Collar performance

The GPS collars used in my study performed satisfactorily, collecting a large amount data over a relatively short period. To avoid leopards removing their collars, I would recommend attaching collars slightly tighter, leaving only enough space for three fingers to slide under the collar. However, at the end of the required tracking period it would be imperative to recapture these collared animals and remove the collars, to prevent any long-term adverse effects. Another important factor influencing the length of the study is the battery life of the collars, which is influenced by the number of GPS fixes that the collar logs per day, as well as the upload intervals. Ideally, the more GPS fixes one collects, the higher the resolution of the data. Therefore, it may be worth sourcing collars with the longest guaranteed battery life. Downloading of collars is an important factor to consider, as, in order for the UHF receiver to communicate with the collars, I had to be within 300m of the leopards. This is often dangerous and should be avoided. If one's study area has cellular network coverage, I would recommend GSM/GPS collars, where the data from the collars is downloaded via a cellular network signal whenever the animals are within the network range.

5.2 Management recommendations

5.2.1 Sustainable off-take quotas for the Mangwe area

Based on the density estimates of this study, I extrapolated the results from my study area, to the area previously delineated by the Marula ICA (Figure 2.3) within the Mangwe region, in order to provide a rough estimate for the entire area. This boundary was suitable for the purposes of this extrapolation, as previous quota data reported by Purchase (2006) apply to the former Marula ICA boundaries, and the landuse patterns and habitats are similar. Using a minimum leopard population estimate of 4 leopards/100 km^2 and a maximum of 6 leopards/100 km^2 (based on the camera-trapping results), I calculated a minimum leopard population estimate of 114 and a maximum of 171 for the region. For visual representation purposes, I estimated the number of males, based on an average population density of 5 leopards/100 km^2, using the ratio of 1 male: 1.6 females, which resulted in an average of 55 males (including sub-adults) for the former Marula ICA area (Figure 5.1). This illustrates the total number of males estimated for the area and the proportion of males that can be sustainably removed, as only males can be shot legally as trophies in Zimbabwe (Purchase 2006).

Based on these estimates, I calculated a maximum and minimum sustainable off-take, based on either 8% (Figure 5.1.b & c) or 4% (Figure 5.1.d & e) of the total leopard population. It is unclear which of these off-take percentages the ZPWMAs base their annual leopard off-take quotas on. Purchase (2006) reports that the off-take quotas are based on 8% of the total estimated leopard population of an area. However, the ZPWMA senior ecologist mentioned that since 2010, off-take quotas were based on 4% of the total leopard populations (G Mtare 2011 pers. comm.). Based on an off-take model investigating the sustainability of hunting on big game populations, it appears that a maximum sustainable off-take of 3.8% is recommended for leopards, except where females are also included in incidental off-take. In this case, a sustainable off-take of 3.6% is recommended (Caro *et al.* 2009). The Safari Operators Association of Zimbabwe (SOAZ) has acknowledged that female leopards are occasionally shot as trophies (Purchase 2006), and an anecdotal assessment of 10 skins exported from western Zimbabwe, found that four of these were females, and two were sub-adult males (Anon 2009). Therefore, I would strongly recommend that managers adopt the precautionary principle, and apply the sustainable off-take at 3.6% rather than 8%. I would also recommend that the minimum estimate of five quota be allocated as the quota for the entire Mangwe area (Figure 5.1.e).
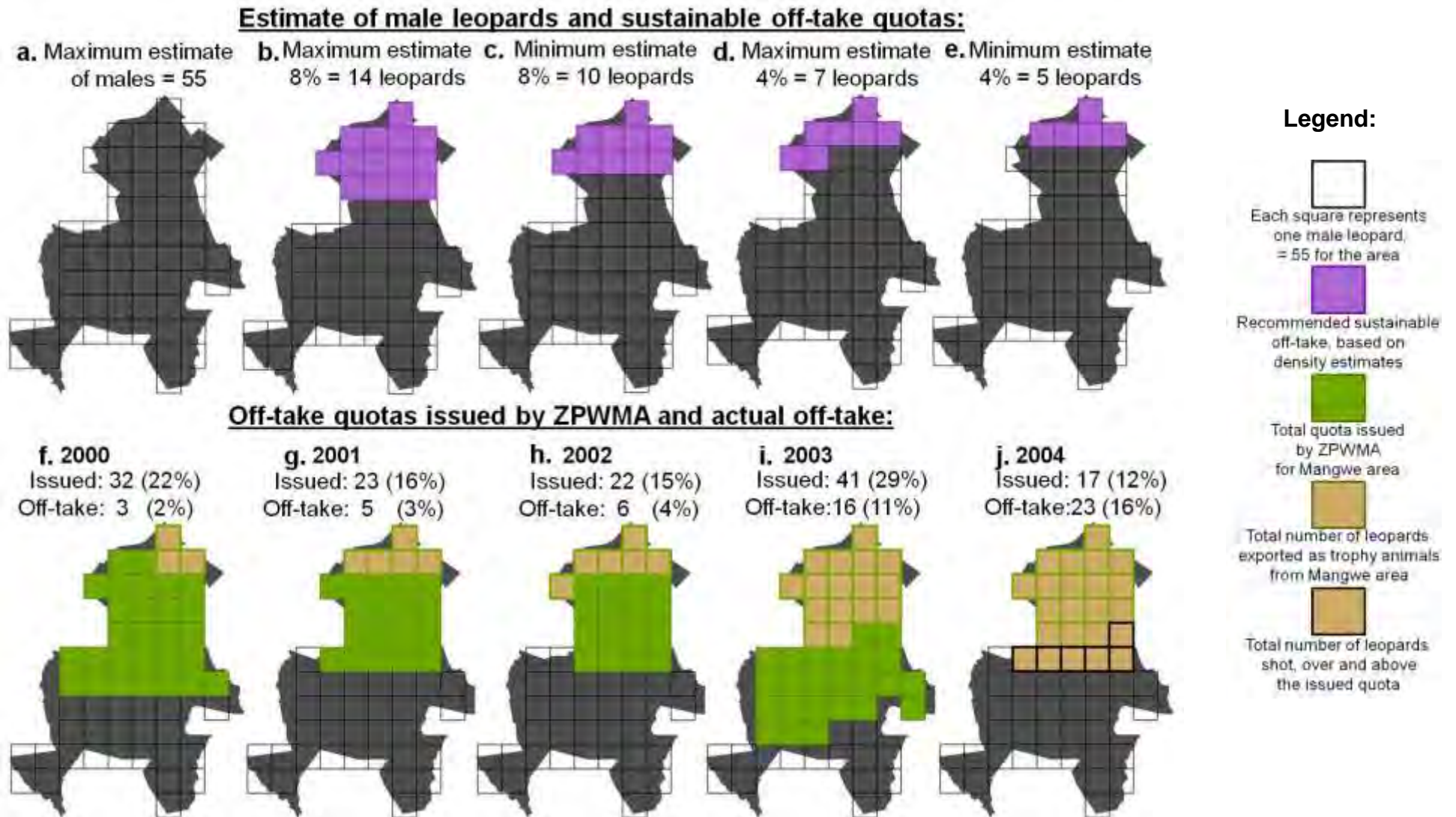
Figure 5.1: Insets show the Mangwe area boundary (former Marula ICA boundary), with an extrapolated grid overlaid. This grid represents the total estimated male leopards (*n*=55) for the area, based on the average recommended camera-trapping density estimate of 5 leopards/100 km², and the ratio of 1 male: 1.6 females from these results. The recommended sustainable off-take for leopards is 4%, although ZPWMA have been using 8%. The off-take quotas from 2000 – 2004 are based on Purchase's (2006) report, with the percentage (in brackets) of the total estimated population (as per my average estimated total population figure) that this quota represents.

Based on this brief assessment of the sustainability of the Mangwe quotas from 2000 to 2004 (Figure 5.1), there appears to be some discrepancy in the number of quotas issued each year, almost doubling from 2002 (22) to 2003 (41) and then dropping again in 2004 (17). In 2004, it appears that six additional leopards were killed, over and above the number of quotas issued for that year. In 2000, only three leopards were reported to have been shot in the area. This could have been the result of the drop in tourism that year, in response to the civil unrest in the country, reducing the number of trophy hunts (Grant 2008). Although this is likely to have been the case in 2000, in 2001, the number of leopards reported to have been shot was five. However, one hunting operator reported killing eight leopards as trophies in that year (Grant 2008). There is a recognised limitation in the ZPWMA's recording of the actual number of leopards killed annually, therefore it is difficult to accurately assess this off-take (Purchase 2006). So, although it appears as if the off-take was sustainable, in comparison to the quotas issued from 2000 to 2002, it is likely that these off-take figures have been underestimated.

Based on my average population density estimate of 5 leopards/100 km^2, the quotas issued by the ZPWMA represented 22% of the total leopard population in 2000, 16% in 2001, 15% in 2002, 41% in 2003 and 12% in 2004. These are all much higher than the recommended sustainable off-take percentage of 4% and also higher than the 8% that the ZPWMA have been using in the past (Purchase 2006). Therefore, previous population estimates, upon which these sustainable off-take quotas have been based, must have been highly overestimated.

5.2.2 Trophy hunting management recommendations

The brief assessment above highlights many of the issues that need to be addressed if leopards are to be hunted sustainably in Zimbabwe. These are:

- Hunting quotas need to be based on current, robust density estimates, where these are available.

- The population estimates based on Martin and de Meulenaer's (1988) model need to be revisited, and, based on their projections, these estimates should have been halved in 2008 due to the inevitable increase in human population and the resultant decrease in natural habitats. Therefore, until the National Leopard Management Project manages to produce a robust national estimate, quotas should be reduced to 250 leopards/ year for Zimbabwe (Purchase 2006).

- Quotas should be distributed based on ecological boundaries, rather than administrative farm boundaries. One quota should be issued to an area, and then individual permits distributed based on male leopard home range sizes within this area. Bailey (1993) suggests that areas

where leopards are trophy-hunted and a male is removed, no other males should be hunted within that home range for at least two years, allowing for the successful survival of at least one litter of cubs. Therefore, quotas could be distributed based on adult home range sizes, and rotated each year, determined by where leopards were shot in previous years. Balme *et al.* (2010a) recommend the management of areas as leopard hunting zones (LHZ) - areas where leopard populations are dense and robust enough to sustain annual hunting. They recommend an area of about 600 km^2, from which one leopard can be hunted annually (Balme *et al.* 2010a). Within this LHZ, the same property should not be able to hunt a leopard in consecutive years, thus making allowance for Bailey's (1993) recommendation for cub survival.

- Better monitoring and reporting is needed in order to flag any trends in population size, trophy size or population structure. Any decreasing trends, or departures from the 'norm' could signal the need for adaptive management practices before it is too late (du Toit 2002). A recent study in Zimbabwe attempted to assess the trends in leopard skull sizes over time (Sibanda 2011). A significant decrease in leopard trophy size was recorded over four years. However, the sample size in this study was very low (Sibanda 2011). The anecdotal assessment of 10 skins, mentioned above, could be an indication that adult male leopards are declining, increasing the likelihood of shooting females or sub-adult males as trophies (Anon 2009). These trends need to be recorded and assessed. Packer *et al.* (2011) suggest that any decline in harvest is likely to reflect a declining population size. An independent body, trained to measure trophy sizes and sex leopard skins, would be needed to accurately record trophy data and the location of where these trophies originated. The ZPWMA admit to not having the capacity to deal with such requirements (G Mtare pers. comm.) and therefore an independent body should be employed to deal with this aspect of trophy hunting. Standardised data forms would be needed, and DNA samples such as teeth or hair could be collected from each trophy for future research and monitoring purposes.

- The effects of hunting females as trophy animals needs to be investigated and acted upon. Caro *et al.*'s (2009) model and Balme *et al.*'s (2010) assessment of leopard in KwaZulu-Natal, South Africa both show that the hunting of females can have adverse effects on leopard populations. However, a simulation model run by Packer *et al.* (2009) suggests that cougar populations can theoretically withstand higher levels of harvesting of females than males, as a male-only harvest can have adverse effects on the viability of the population of a polygynous species with sexually-selected infanticide (Packer *et al.* 2011). Based on this pattern, it would be worth investigating whether this theory would be applicable to leopards in Zimbabwe, or if more should be done to stop this illegal hunting of females because of the possible adverse effects on the survival of the leopard populations. Therefore, based on past

leopard research, I would not recommend any female off-take until further information is available.

- Trophy hunting alone is unlikely to destroy leopard populations in Zimbabwe. However, trophy hunting, along with unregulated hunting (by landowners), poaching, problem animal control, natural prey depletion, the demand for leopard skins for cultural and religious beliefs, and habitat loss, could have serious deleterious effects on leopard populations if not managed appropriately. Thus, a multi-disciplinary approach to leopard research and management should be undertaken immediately.

5.3 Recommendations for future research

One of the main limitations of this study was the small sample size of collared leopards. The territorial behaviour and home range overlap amongst males would be key factors to investigate in order to assess the effects of trophy hunting. A long-term study of the movement behaviour of a number of collared males and females would therefore be beneficial for leopard conservation. Incorporated in such a study, one could also assess habitat use and avoidance behaviour. No habitat mapping data is available for the Mangwe area, nor any high resolution or recent aerial photographs. This would further benefit any extensive leopard research in the area.

An investigation into the source/sink dynamics between the neighbouring Matopos National Park to the east and the Mangwe area would be an interesting assessment. The Mangwe area has been heavily hunted for the last 20 years, yet only in the last six years is a decline in the number of adult males being noticed. It would be beneficial to the survival of these leopard populations to investigate whether these potential sink effects are having any adverse effects on the leopard population within the national park.

Another very important study for the Mangwe area would be an assessment of the human-carnivore conflict in the area. Investigating the perceptions and attitudes of the landowners in the area, as well as estimating their financial loss would be vital for any further human-wildlife conflict management. It has also been recently reported that leopards are wrongfully persecuted for killing livestock, when the culprits are often spotted hyena. An investigation into these problems is important in order to assess problem animal control and carnivore conservation in the area.

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APPENDICES

Appendix I: List of scientific names of all mammal species mentioned in text

Common name	Scientific Name
Black rhino	*Diceros bicornis*
Brown hyaena	*Parahyaena brunnea*
Cattle	*Bos domesticus*
Cheetah	*Acinonyx jubatus*
Domestic dogs	*Canis familiaris*
Donkeys	*Equus asinus*
Elephant	*Loxodonta africana*
Humans	*Homo sapiens sapiens*
Jaguar	*Panthera onca*
Leopard	*Panthera pardus*
Lion	*Panther leo*
Ocelot	*Leopardus pardalis*
Rock hyrax	*Procavia capensis*
Snow leopard	*Uncia uncia*
Spotted hyaena	*Crocuta crocuta*
Tiger	*Panthera tigris*
White rhino	*Ceratotherium simum*
Yellow-spotted rock hyrax	*Heterohyrax brucei*

Appendix II: List of species (family, scientific name and common name) detected during camera survey showing the total number of photographs taken per species, their percentage (%) contribution to the total number of photographs taken and their capture frequency (number of photographs/100 camera days).

Family	Species	Common name	*n*	Spp. %	CF	Effort
Anthropogenic	*Homo sapiens sapiens*	Humans	88	17.4	11.0	N/A
	Canis familiaris	Domestic dogs	7	1.4	0.9	N/A
	Bos domesticus	Cattle	164	32.3	20.5	N/A
	Equus asinus	Donkeys	10	2.0	1.3	N/A
Canidae	*Canis mesomelas*	Black-backed jackal	9	1.8	1.1	265
Bovidae	*Tragelaphus oryx*	Eland	1	0.2	0.1	2395
	Tragelaphus strepsiceros	Greater Kudu	8	1.6	1.0	298
	Aepyceros melampus	Impala	25	4.9	3.1	94
	Connochaetes taurinus	Blue wildebeest	5	1.0	0.6	478
	Oretragus oretragus	Klipspringer	12	2.4	1.5	198
	Raphicerus campestris	Steenbok	6	1.2	0.8	398
	Sylvicapra grimmia	Common (Grey) duiker	19	3.7	2.4	125
Cercopithecidae	*Cercopithecus pygerythrus*	Vervet monkey	3	0.6	0.4	797
	Papio hamadryas	Chacma baboon	19	3.7	2.4	125
Equidae	*Equus burchelli*	Burchell's Zebra	4	0.8	0.5	598
Felidae	*Caracal caracal*	Caracal	1	0.2	0.1	2395
	Felis lybica	African wildcat	10	2.0	1.3	238
	Panthera pardus	Leopard	6	1.2	0.8	398
Galagidae	*Galago moholi*	Lesser bushbaby	1	0.2	0.1	2395
Herpestidae	*Galerella sanguinea*	Slender mongoose	1	0.2	0.1	22
	Mungos mungo	Banded mongoose	3	0.6	0.4	797
	Ichneumia albicauda	White-tailed mongoose	7	1.4	0.9	341
Hyaenidae	*Crocuta crocuta*	Spotted hyaena	4	0.8	0.5	598
	Parahyaena brunnea	Brown hyaena	20	3.9	2.5	118
Hystricidae	*Hystrix africaeaustralis*	Porcupine	14	2.8	1.8	170
Mustelidae	*Mellivora capensis*	Honey badger	1	0.2	0.1	2395
Orycteropodidae	*Orycteropus afer*	Antbear (Aardvark)	11	2.2	1.4	216
Pedetidae	*Pedetes capensis*	Springhare	1	0.2	0.1	2395
Rhinolophidae	*Rhinolophus sp.*	Horseshoe bat	1	0.2	0.1	2395
Sciuridae	*Paraxerus cepapi*	Tree squirrel	1	0.2	0.1	2395
Suidae	*Potamochoerus porcus*	Bushpig	7	1.4	0.9	341
	Phacochoerus africanus	Warthog	1	0.2	0.1	2395
Viverridae	*Civettictis civetta*	African civet	2	0.4	0.3	1197
	Genetta tigrina	Rusty spotted genet	8	1.6	1.0	298
Mammals	Unidentifiable	Unidentifiable	3	0.6	0.4	N/A
Reptiles	*Gerrhosaurus validus*	Giant plated lizard	1	0.2	0.1	N/A
Birds	*Bucorvus leadbeateri*	Southern Ground-Hornbill	1	0.2	0.1	N/A
	Buphagus erythrorhynchus	Red-Billed Oxpecker	3	0.6	0.4	N/A
	Caprimulgus sp.	Nightjar	1	0.2	0.1	N/A
	Lamprotornis nitens	Cape Glossy Starling	1	0.2	0.1	N/A
	Numida meleagris	Helmeted Guineafowl	2	0.4	0.3	N/A
	Prionops retzii	Ret's Helmet-Shrike	1	0.2	0.1	N/A
	Pternistis natalensis	Natal Spurfowl	8	1.6	1.0	N/A
	Pternistis swainsonii	Swainson's Spurfowl	1	0.2	0.1	N/A
	Tockus erythrorhynchus	Red-billed Hornbill	2	0.4	0.3	N/A
	Tockus leucomelas	Southern Yellow-Billed Hornbill	1	0.2	0.1	N/A
	Turdus libonyana	Kurrichane Thrush	1	0.2	0.1	N/A
	Unidentifiable	Unidentifiable	1	0.2	0.1	N/A
TOTAL			507			

Appendix III: Capture matrix of leopards captured in the baited camera-trapping study, used for analysis in CAPTURE. Rows A-M representing individual leopards and columns 1-58 representing the number of sampling occasions. 1 indicates a captures and 0 no capture.

```
    1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39 40 41 42 43 44 45 46 47 48 49 50 51 52 53 54 55 56 57 58

A   0 0 0 0 0 1 0 0 0 0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  1  1  0
B   0 0 0 0 0 0 0 0 0 0  0  0  0  0  0  0  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  1  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
C   0 0 0 1 1 1 0 1 0 1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  1  0  0  1  0  1  0  0  0  0  0  0  0  0  0  0  0  0  0
D   0 0 0 0 0 0 0 0 0 0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  1  0  0  0  1
E   0 0 0 0 0 0 0 0 0 0  0  0  1  0  0  0  1  0  0  0  0  0  0  1  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  1  0  0  0  0  0  1  0  0  0
F   0 0 0 1 0 0 0 0 1 0  0  0  0  0  0  1  0  0  0  0  0  0  1  1  1  0  1  1  1  0  1  1  1  1  1  1  1  0  0  0  0  0  0  0  0  0  0  0  1  0  0  1  1  1  1  1  0  0
G   0 0 0 0 1 1 0 0 0 0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
H   1 0 0 0 0 0 0 1 0 0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
I   0 0 0 0 0 0 0 1 0 0  0  0  0  0  0  0  0  0  0  0  0  0  0  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
J   0 0 0 0 1 1 0 1 1 0  0  1  0  1  0  1  1  1  1  1  1  0  0  1  1  0  0  0  0  0  1  1  1  0  1  0  1  1  0  1  0  1  0  1  0  1  1  1  0  0  0  0  0  1  1  1  0  0
K   0 0 1 0 1 1 0 0 0 0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
L   0 0 0 0 0 0 0 0 0 0  0  0  0  1  0  0  0  0  1  1  0  0  0  0  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
M   0 0 0 0 0 0 0 1 1 0  0  1  0  0  0  1  1  0  0  0  0  0  1  1  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  1  0  0  0  0  1  0  0  1  0  0  1  0  0  1  0  0  0
```

Appendix IV: Example of SPACECAP 'Animal Capture Details' file

Loc_ID	Animal_ID	SO
19	8	8
1	11	10
8	3	11
11	6	11
8	3	12
1	7	12
15	10	12
1	11	12
1	1	13
8	3	13
1	7	13
15	10	13
1	11	13
8	3	15
19	8	15
19	9	15
17	13	15
17	10	15
13	6	16
14	10	16
17	13	16
6	3	17
16	13	19
16	5	19
15	10	19
14	10	21
20	12	21
17	2	23
16	5	23
11	6	23
15	10	23
16	13	23
20	10	24
16	13	24
14	10	25
14	10	26
20	12	26
20	10	27
20	12	27
8	6	30
15	10	30
17	13	30
17	5	31
13	6	31
14	10	31
17	13	31
17	5	32
13	6	32
7	9	32

Appendix V: Example of SPACECAP 'Trap Deployment Details' file. A 1 represents when the camera-traps were functioning and a 0 when they were not.

Loc_ID	X_Coord	Y_Coord	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1	610878	7702614	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
2	613645	7702217	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
3	613159	7700094	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
4	610046	7699602	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
5	608528	7695990	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
6	611041	7694950	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
7	614442	7697527	1	1	1	1	1	0	1	1	1	1	1	1	1	1	1	1
8	612972	7695332	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
9	616873	7695977	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
10	610007	7693303	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
11	614775	7694022	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
12	615456	7689609	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
13	611851	7692347	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
14	608846	7690240	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
15	609578	7687637	1	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1
16	612627	7690532	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
17	615158	7685989	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1
18	613667	7686864	1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1
19	612552	7697382	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
20	609639	7684724	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Appendix VI: Detailed description of how Home Range Centres were created

'State-space' boundaries based on:
- HMMDM buffer of 2.5 km
- MMDMOSA buffer of 3.9 km
- MDM buffer of 12 km

Potential home range centers:
- 0: Unsuitable habitat
- 1: Suitable habitat

- Camera trap sites
- 500m x 500m grid mesh
- Minumum area rectangle
- Study area boundary
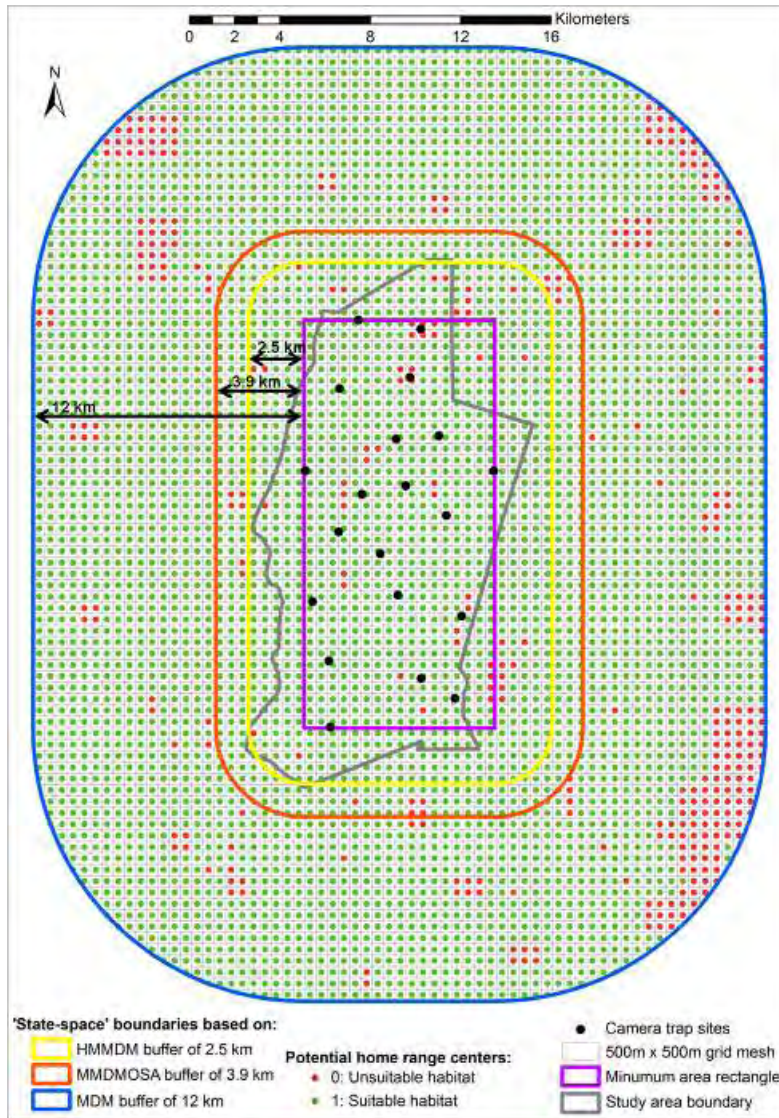
Make sure all spatial data for CAPTURE and SPACECAP have been converted to UTM co-ordinates.

A simple way of changing Decimal Degrees to UTM in ArcGIS:
Once you have imported your shapefile, open the attributes table, create new fields for your x and y co-ords, make sure you define them as a long integer, then you start editing the shapefile – go to that new field you have created, right click -> calculate geometry , make sure the property tab has the correct co-ordinate (i.e. x for Easting and Y for Southing), select the drop down list for units and select meters, then do the same for the next co-ordinate field, this will calculate UTMs for all your exiting co-ordinates.

To create home range centres:
Take the outer camera sites and draw a rectangle around them (Purple rectangle). Then buffer this rectangle – to include any overlapping home range areas of any leopards photographed whose home range may lie partly outside the boundary. Create a grid 500 x 500m – and place a dot at the centre of each grid square – you can do this all with Hawth's tools. By creating a dot in the middle of each grid square you are creating co-ordinates for each, they must be in UTM. You then have to assess each grid as to whether a leopard could exist there or not – if yes, in the attributes table, put a 1 next to that co-ord, if no put a 0. So if there are villages or a dam etc then it would 0. This then gives you your data for the last input file: Potential_HR_Centres.

Appendix VII: Example of SPACECAP 'Home Range Centres' file. A 1 represents potential leopard habitat and a 0 represents unsuitable leopard habitat

X_COORD	Y_COORD	HABITAT
606263	7704984	1
606763	7704984	1
607263	7704984	1
607763	7704984	1
608263	7704984	0
608763	7704984	1
609263	7704984	1
609763	7704984	1
610263	7704984	1
610763	7704984	1
611263	7704984	1
611763	7704984	1
612263	7704984	1
612763	7704984	1
613263	7704984	1
613763	7704984	1
614263	7704984	1
614763	7704984	1
615263	7704984	1
615763	7704984	1
616263	7704984	1
616763	7704984	1
617263	7704984	1
617763	7704984	1
618263	7704984	1
618763	7704984	1
619263	7704984	1
606263	7704484	1
606763	7704484	1
607263	7704484	1
607763	7704484	1
608263	7704484	1
608763	7704484	1
609263	7704484	1
609763	7704484	1
610263	7704484	1
610763	7704484	1
611263	7704484	1
611763	7704484	1
612263	7704484	1
612763	7704484	1
613263	7704484	1
613763	7704484	1
614263	7704484	0
614763	7704484	0
615263	7704484	1
615763	7704484	1
616263	7704484	1
616763	7704484	1

Appendix VIII: Morphological data sheet (See next appendix for details on the   grey highlighted sections).

Date:		GPS:	S(-)		Time:
Leopard ID:			E		Temp:
Capture Site:					
Capture Method:					
Bait used:					
Collar number:			Radio frequency:		
Immobilisation:					
Drug:			Dosage:		
Dart in time:			Dart site on animal:		
Top ups:			Animal down time:		
Dose:			Time of recovery:		
Sex:		M		Body Condition:	
		F		Demeanour:	
Age: +/- 5 years		Adult			
		Subadult			
Sample checklist:			**Photo checklist:**		
Tissue			Face		
Hair			Teeth		
Blood slide			Tail		
Faeces - formalin			Sides (for camera-traps)		
Faeces - ethanol					
Ectoparasites					
Cheek scrape					
Measurements:					
Weight		kg	Chest girth		cm
Tail length (**B-C**)		cm	Chest depth (from back to chest)		cm
Nose to tip of tail length (**A-D**)		cm	Neck girth		cm
Stomach size:		cm	Shoulder height (leg bent)		cm
Skull width:		cm	Shoulder height (full length)		cm
Skull length:		cm			
Canines:					
Top left canine(from gum, side)		cm	Top right canine		cm
Bottom left canine		cm	Bottom right canine		cm
Distance between top canines		cm	Distance between lower canines		cm
Front right paw:			*Back right paw:*		
Total length		cm	Total length		cm
Total width		cm	Total width		cm
Pad length		cm	Pad length		cm
Pad width		cm	Pad width		cm
Distinguishable characteristics and comments:					

Appendix IX: Morphological measurements guide (adapted from by Balme et al. 2011)
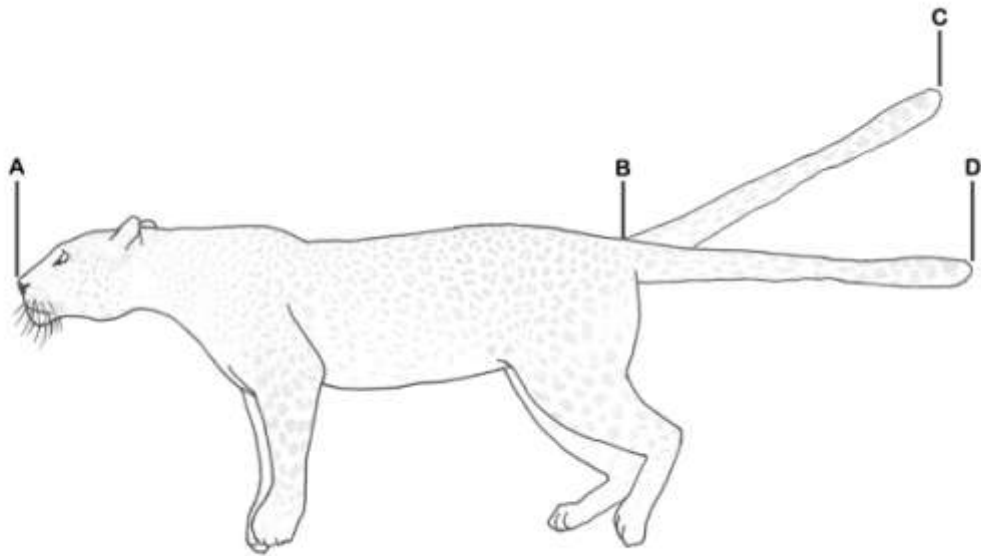
These measurements correspond with the previous Appendix (Morphological data sheet), specifically the highlighted sections. Measurement from A to D would be for nose to tip of tail, and B to C would be the tail length.

Appendix X: AWT GPS collar battery life estimation

- Number of GPS readings per day - 2
- Number of data uploads per day - 288
approximately 943.656 days / 2.59 year

- Number of GPS readings per day - 2
- Number of data uploads per day - 144
approximately 1287.2 days / 3.53 year

- Number of GPS readings per day - 4
- Number of data uploads per day - 288
approximately 813.128 days / 2.23 year

- Number of GPS readings per day - 4
- Number of data uploads per day - 144
approximately 1055.984 days / 2.89 year

- Number of GPS readings per day - 6
- Number of data uploads per day - 288
approximately 714.328 days / 1.96 year
-every 5 mins

- Number of GPS readings per day - 6
- Number of data uploads per day - 144
approximately 895.176 days / 2.45 year
-every 10 mins