EVALUATION OF SPOOR TRACKING TO MONITOR CHEETAH ABUNDANCE IN CENTRAL NORTHERN NAMIBIA.

by
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PREFACE

The research described in this mini-dissertation
was carried out at the Centre for Environment,
Agriculture and Development, University of KwaZulu-Natal,
Pietermaritzburg, under the supervision of Prof. Michelle Hamer.

This mini-dissertation represents the original work
of the author and has not otherwise been submitted in any form
for any other degree or diploma at any university.
Where use has been made of the work of others
it is duly acknowledged in the text.

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ABSTRACT

The design, implementation, management and the evaluation of sound conservation practices, is often dependent on the availability of reliable estimates of animal abundance. Large carnivores often pose particular problems in this regard, due to their low densities and wide-ranging behaviour, so the true abundance of such species are seldom able to be reported in literature. As a result, the use of indices of abundance, mostly for relative abundance, has been investigated. However, before these indices can be reliably utilized for conservation purposes, there is a pressing need to calibrate them.

As of yet, calibration studies have primarily been performed on demarcated conservation areas, where individuals could be individually identified. Not all these calibrations studies reported indices to be a function of true density. Nevertheless, spoor frequency has been reported to be a function of true density for carnivores in certain Parks in Namibia. Precisely, cheetah spoor density was reported to correlate with visuals in the Kgalagadi Transfrontier Park. The majority of these studies elucidate a species spatial organization, animal behaviour, as the paramount factor determining the relationship between densities estimated via different censusing methods. Thus, the efficiency of spoor frequency to estimate and monitor relative abundance for wild cheetahs is yet to be empirically tested.

Despite the lack of a true density estimate for the free-ranging cheetahs in the study area, evaluated spoor tracking as a possible index to monitor relative cheetah abundance using radio-telemetry densities estimates as representative of true abundance for the area, for the 1995 to 2000 period. The study is considered to be opportunistic, and a pillar for future research, as transects where spoor tracking was conducted were layout primarily for ungulates strip counts. Least-linear regression and Spearman's correlation were used to evaluate the relationship between density estimates derived by the two methods. Percentages of change on annual densities were also regressed as a mean to test spoor frequency sensitivity to density changes.

The calibration of spoor frequency with estimates of density produced using radio-telemetry, without the ascription of imprints to individual animals, was poor (r_s=17.4, y=0.36+0.20). The sensitivity analysis also showed spoor tracking poor reliability to monitor cheetah population. This can be attributed, in order of importance, to the discrepancies on the spatial extent sampled by the two methods, the species large home ranges, substrate quality, habitat preferences, the availability of farm road networks and the transect design, i.e., cyclic. However, the paramount factor limiting the study conclusions was the lack of a more local density estimate at a farm level. Therefore, the use of spoor frequency to estimate wild cheetah relative abundance requires further research, particularly using a different sampling design, longer straight transects and the acquisition of local densities estimates.

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Chapter 1

Chapter 1

GENERAL INTRODUCTION TO THE STUDY

"The difficulty of obtaining accurate counts of the number of individuals...has led to attempts to develop indices of abundance for the species concerned... it is sufficient to know the relative abundance of a particular species in different areas or at different times without having an exact count of the population. " Dice 1941:402)

1.1 Introduction

The earth has been in a dynamic state of continuous motion and change as far back as its existence is dated, which has effected its inhabitants' distribution and abundance. It was the continuous movements and collisions of tectonic plates that allowed for the migration and colonization of new areas by mammalian carnivores (Bothma & Walker 1999), including the cheetah (Adams 1979).

Carnivores in general and cheetah in particular, once had a wider range and higher abundance than at present (Hunter 2001). Both the range and abundance of cheetah have decreased due to a number of factors, resulting in their current conservation status as threatened (Nowell 2002; Ray, Hunter & Zigouris 2005). Effective conservation practices are dependent on reliable estimates of population demographics. Furthermore, the monitoring of carnivores' relative abundance has become a priority (Loveridge, Lynam & Macdonald 2001; Pollock, Nichols, Simons, Farnsworth, Bailey & Sauer 2002), and this requires the use of cost-effective, repeatable, and easily implementable methods. The determination of true densities for carnivores it is not always necessary for management purposes (Dice 1941).

The refinement of density estimate techniques, including the use of signs of animal presence in an area to derive relative density estimates, has been and continues to be explored as the costs of employing more formal methods, such as mark-recapture or radio telemetry remain high. In addition, environmental and design factors may introduce bias to population estimates obtained using these methods. Evaluation of indices of population density is essential and should consider effort, efficiency, and / or adequacy and performance (Suchman 1967) and should be calibrated prior to implementing population estimate techniques in a monitoring scheme (Eberhardt & Simmons 1987). Calibrations are, however, seldom conducted (Walker, Pancotto, Schachter-Broide, Ackermann & Novaro 2000; Gese 2001) primarily due to the lack of true density estimates.

1.2 Study rationale

Both the range and abundance of carnivores have declined due to a number of factors. The implementation of sound conservation measures to ensure the survival of carnivores relies on accurate estimates of population sizes, whether these are absolute or relative. There are currently no generally accepted methods for estimating population density for cheetah mainly because of a combination of aspects of the behavioural ecology of this species which make counting difficult and the prohibitive costs of counting individuals. Therefore, a need exists for employing reliable, repeatable, cost-effective and easily implemented methods to monitor populations of cheetah. Spoor tracking has been identified as a feasible non-invasive census technique for monitoring carnivores. This technique has been commonly used for a range of carnivores but mostly in conservation areas, and its applicability to wild, free-ranging populations is poorly understood. Furthermore, spoor tracking has not been evaluated against a true density estimate for reliability purposes for cheetah. This is the focus of the study presented here.

1.3 Aim and research question

The aim of this study is to provide a greater insight into the precision and accuracy of census methods for providing population estimates for cheetah. The study attempts to answer the following research question:

- ✓ Does spoor density estimated using spoor frequency, without visual identification of the individual to which an imprint belongs, show any relationship with population density as estimated by radio telemetry?

The Namibian cheetah subspecies, *Acinonyx jubatus jubatus*, is the subspecies considered primarily in this dissertation, but results may be applicable to a wider range of large carnivores.

1.3.1 Objectives

In order to attain the stated aim four objectives are identified:

1. To determine cheetah density per unit of effort (kilometers) and area (square kilometers) based on spoor frequency, using the spoor tracking censusing method.
2. To investigate the relationship between cheetah densities measured from two indirect census techniques (spoor tracking and radio telemetry).
3. To determine the optimum sampling effort required to obtain reliable spoor frequency counts (determine sampling effort in kilometers required to minimize variability in counts).
4. To determine the sensitivity of the spoor frequency census method to actual annual changes in cheetah density as measured by radio telemetry.

1.4 Thesis layout

The first part of the thesis (Component A) has five chapters, including this one. Chapter two presents the conceptual framework and the approach for the study. Chapter three provides a background overview on cheetah with an emphasis on its ecology. Chapter four focuses on current and previous census techniques used for monitoring wildlife, with an emphasis on adequacy to derive estimates of animal abundance. Both general techniques for carnivores and specific ones for cheetah are reviewed. Indirect indices are also discussed in this chapter. The study methodology is presented in Chapter five. In the second part of the thesis (Component B) the study is presented as a scientific manuscript in the format of the South African Journal of Wildlife Research.

Chapter 2

CONCEPTUAL FRAMEWORK

2.1 Introduction

The Millennium Ecosystem Assessment (2003) definition of a well designed conceptual framework for different purposes is as follows:

> "A well-designed framework for either assessment or action provides a logical structure for evaluating the system, ensures that the essential components of the system are addressed as well as the relationships among those components, gives appropriate weight to the different components of the system, and highlights important assumptions and gaps in understanding."

This definition provided the foundation for the conceptual framework of this study. Therefore, the aim of this chapter is to provide a description of the different areas of literature reviewed and to establish how these are integrated relative to the topic investigated in this study and its aim and objectives.

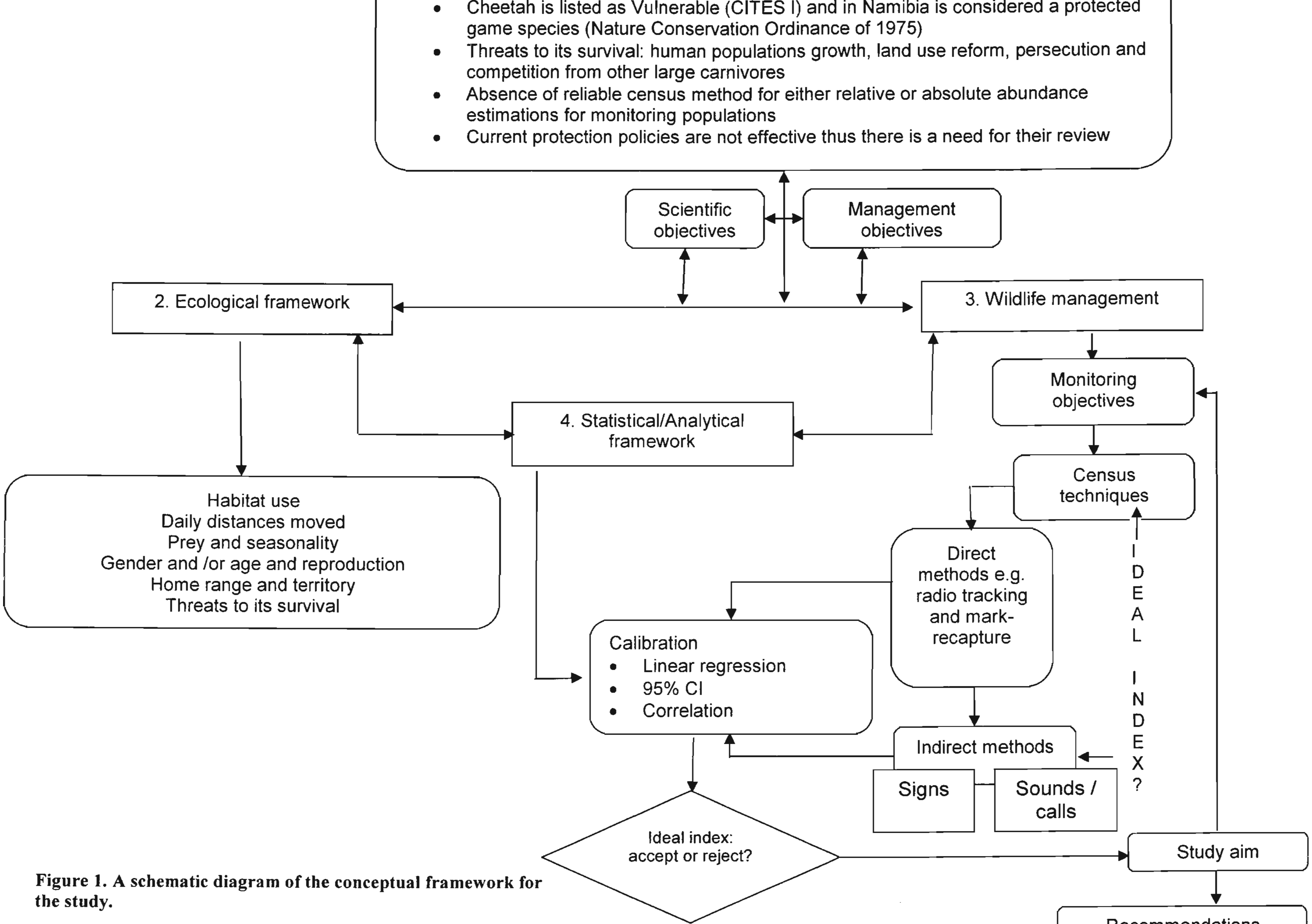

Figure 1. A schematic diagram of the conceptual framework for the study.

2.2 Framework description

2.2.1 Description of components

Four major subcomponents that inform the study, i.e. a system, were identified. The situational analysis subcomponent provides: (1) the background of the current species status and (2) it highlights the need for further research in respect to finding a census method that is repeatable, relatively cost-effective, and easy to implement or that can be integrated with other practices, but provides reliable relative or absolute abundance estimates.

The ecological framework, the second subcomponent, in turn, focuses on the key ecological aspects of the species that give merit to use of indices of abundance, such as spoor tracking, as well as those that affect the design and implementation of such methods.

The third sub-component is the management of wildlife. This in turn will provide the necessary theoretical and empirical background on the different census techniques employed to date to estimate carnivore, particularly cheetah, abundance. However, due to similarities in terms of factors that influence sign indices either positively or negatively, and which may result in doubts regarding the reliability of derived estimates, reference to other studies besides carnivores is made. This also highlights issues related to the design of census programs.

Management and scientific objectives constitute the interface layer between the situational analysis and both the ecological and wildlife management subcomponents. There is a cyclic flow of information between this interface and the subcomponents. For example wildlife policy formulation relies on accurate and reliable information from the subcomponents, so that the objectives are successfully accomplished. The fourth and last subcomponent, the statistical or analytical framework, presents the necessary tools to test the assumptions of whether an index is indeed ideal. These are tools such as univariate analysis, i.e. calibration or the regression of an index of abundance against another estimate assumed to be a more accurate estimate of true density. The acceptance or rejection of the null hypothesis, that the index (i.e. spoor frequency) is indeed an ideal index for monitoring cheetah wild populations is the primarily focus of the study.

2.2.2 Framework assumptions and research approach

The study is based on the assumption that radio telemetry density is an accurate estimate of true abundance for cheetah in the study area, and that possible factors causing bias are addressed during the analyses. Thus, it is acknowledged that other animals may have made use of the area during the period of data collection and thus encountered spoor were not necessarily those of radio collared animals. On the other hand, spoor tracking is assumed to be an apt index to monitor cheetah abundance based on a review of the literature. Therefore, this study could be considered to be empirical and exploratory (Wisher 2001) (i.e. quantitative), with an emphasis on the testing of a specific hypothesis (Suchman 1967) using primary data (Wisker 2001). It was positivist (Wisker 2001) as it looked at the relationship between the densities estimated via different techniques, with both retroductive and deductive reasoning employed during interpretation of results (Mauton 2001).

Furthermore, it was evaluative research, as it primarily focused on testing the applicability of knowledge and not on its discovery (Suchman 1967). This type of research applies scientific methods to determine the usefulness of a method, with the end result of aiding with planning, designing, developing and monitoring of programs (Suchman 1967). An evaluation of a program can be performed by looking at the program effort, performance, adequacy performance, efficiency and process (Suchman 1967). Here, spoor frequency is evaluated on the basis of efficiency, defined as the accuracy of the method in relation to telemetry densities, as well as effort, defined as the required sampling effort to attain certain precision levels.

2.2.3 Framework and the study results

The end result of this evaluation is to provide an indication of whether spoor frequency under current sampling protocols could be a function of density by comparing it to densities ascertained by radio telemetry, and to make recommendations on how to improve current sampling methodologies as a way of improving the accuracy of future results. Therefore, evaluation here is the determination of whether the results obtained under current sampling protocols, although not designed specifically to answer this study's research question, are of significance or not for designing and implementing spoor frequency as an index for cheetah in Namibia.

2.3 Conclusion

The conceptual framework for the study provided a logical and pragmatic guide for the literature review, the selection of methods, as well as for the development of the aim and corresponding objectives. In this respect, the remaining sections of this dissertation are developed according to the hierarchy displayed in the schematic diagram.

Chapter 3

BACKGROUND HISTORICAL AND ECOLOGICAL ASPECTS OF THE CHEETAH

3.1 Taxonomy and evolution

The cheetah, or 'spotted one' in Hindu (Caro 1994), was first described scientifically by Schreber in 1776 as *Acinonyx jubatus* (Skinner & Smithers 1990). It evolved independently from the other cats of the *Panthera* group (Eaton 1974), splitting about four million years ago (Urbaniak 1998), and originated in North America (Adams 1979). The cheetah has been split into five subspecies (Hunter & Hamman 2003), but the status of the eastern (*A. j. raineyii*) and southern (*A. j. jubatus*) African subspecies is questioned due to their high genetic uniformity (O'Brien, Roelke, Marker, Newman, Winkler, Meltzer, Colly, Evermann, Bush & Wildt 1985).

3.2 Past and present distribution and population estimates

Cheetahs were originally dispersed throughout Africa (Rosevear 1974) and at the beginning of the 19th century Africa harboured a population of approximately 100 000 animals (Myers 1975). Outside Africa, cheetahs were distributed in India, Iran and Afghanistan (Skinner & Smithers 1990). The first recorded cheetah association with man is suggested to have been on the Indian and southern African plains (Bothma & Walker 1999) dating back to the Sumerians 3000 BC (Guggisberg 1975). The relationship between human and cheetah was later described by Eaton (1974) as symbiotic yet utilitarian and problematic.

Towards the end of the 20th century, both abundance and range of the cheetah had constricted. During the mid-1970s, Myers (1975) estimated the continental population to be between 7 000 and 23 000 distributed across 27 countries, and presumed it to be declining. Frame (1984 cited in Nowell & Jackson 1996) estimated a higher upper number (<~25 000) but Marker (2002) advocated a 60 and 65% reduction, respectively on Myers' (1975) and Frame's (1984) upper limits. Hunter (2001) and Marker (2005 *pers. comm.*) both estimated the recent population level to be approximately 12 000 animals.

Outside of Africa, the cheetah was considered to be extinct in India by the 1950's (Ewer 1973; Divyabhanusinh 1995) and only a small population remains in Iran and Afghanistan (Caro 1994; Hunter 2001; Marker 2005 *pers. comm.*). Within Africa the distribution of cheetah has remained fairly stable since the 1970s with two major strongholds, one in eastern Africa (Kenya and Tanzania) and the second in southern Africa (Namibia in conjunction with the Botswana, Zambia and Zimbabwe populations) (Nowell & Jackson 1996; Ray *et al.* 2005). A current estimate for the eastern African subspecies, *A. j. raineyii*, for Tanzania is 600 to 1000 (Gros 2002) and for Kenya the highest potential number is estimated to be 500 to 1000 (Gros 1998). Namibia, on the other hand, lacks a recent estimate and the Morsbach (1986) estimate of 2000 to 3000 is still frequently cited. Primarily based on questionnaires and reports of animal sightings, ongoing research by Stander (2004) indicated a population size of 3138 to 5775 animals countrywide.

Although distribution maps provide an indication of presence, there is a critical need for relative abundance estimates for Namibia and elsewhere, as a lack of these may hamper the implementation of appropriate conservation measures. There is, however, an overwhelming consensus that all carnivore species' abundance has declined in the past, and that this trend is continuing. This fact is prevalent in the literature (Myers 1975 & 1986; Caro 1994; Nowell 1996; Urbaniak 1998; Hunter 2001; Marker 2002; Ray *et al.* 2005), and has resulted in the present conservation status of the cheetah.

3.3 Conservation status of the cheetah

The cheetah is protected both at an international and a national level within most of its range. At an international level it has been listed as Vulnerable on Appendix I of the Convention of International Trade of Endangered Species (CITES) (IUCN 2001; Nowell 2002; IUCN 2005) since the Convention induction in 1975 (Nowell 1996). Appendix I includes species threatened with extinction and therefore any trade either in the form of live export or sport hunting is prohibited except when export and import countries have attained an official permit (Nowell 1996; Stander 2004). Nevertheless, even in the absence of reliable population estimates, Namibia, Botswana and Zimbabwe are permitted by CITES an annual trophy hunting quota of 150, 50 and 5 animals, respectively (Caro 1994; Nowell & Jackson 1996).

At national levels, the cheetah is theoretically fully protected over most of its range (Nowell & Jackson 1996). Namibia has declared the cheetah a protected game species under the Nature Conservation Ordinance of 1975 (Nowell 1996). The system operates on a reporting mechanism whereby farmers register complaints and fates of cheetahs with which they come in contact with to the Ministry of Environment and Tourism (MET) (Nowell 1996).

Protection at either international or national level is based fundamentally on anecdotal and informed guesses of population sizes, re-iterating the need for reliable abundance estimates for the species.

3.4 Reasons for cheetah decline and ongoing threats

The current threatened status of cheetah is due to a number of factors. The first and most longstanding is related to cheetah biology. These cats are only mildly aggressive (Durant 1998), exhibit diurnal hunting patterns, occupy open plains habitats, their speed and hunting prowess made them interesting and appealing to humans (Eaton 1974; Urbaniak 1998), and they are good candidates for domestication. Literature (Eaton 1974; Caro 1994) reveals that as a result of these behaviour patterns, people captured cheetah more than other cats (Hunter 2001). King Abkar the Great collected up to 9000 cheetahs during his lifetime for the purpose of hunting antelope (Divyabhanusinh 1995). Furthermore, Egyptian pharaohs are also recorded to have hunted cheetahs for fur and sport. Such activities are thought to have led to the extinction of cheetah in India and the small remaining population in Iran (Divyabhanusinh 1995). Caro (1994) highlights that at a certain stage East African cheetahs were imported to supplement Asian populations. A factor that exacerbates the impact of hunting and capture on cheetah populations is their relative sociality that allows for the easy removal of entire social groups (Marker, Kraus, Barnett & Hurlbutt 1996).

The arrival of European settlers, who considered cheetahs as vermin (Marker 2002), and the introduction of new technology in the form of guns and poisoning (Hamilton 1986), further increased cheetah removal rates in Africa. Lions (*Panthera leo*) have been exterminated and leopards (*Panthera pardus*), hyenas (*Crocuta crocuta* and *Parahyaena brunnea*) and wild dog (*Lycaon pictus*) numbers have been severely reduced on commercial lands due to poisoning in Namibia (McVittie 1979; Myers 1986; Marker, Mills & MacDonald 2003a). Cheetahs, however, are not threatened by poisoning as they rarely scavenge (Hunter & Hamman 2003). The disappearance of lions and spotted hyenas may have had a positive impact on cheetah populations; according to Laurenson (1994) and Bothma and Walker (1999), as much as 72% of cheetah cub mortality in the Serengeti is due to predation by lions and hyenas.

The second cause of decline and threat to cheetah numbers is human population growth and its corresponding impacts on free-ranging cheetah. Human population growth is accompanied by the conquering of land for residence and subsistence agriculture. The African continent is highly reliant on agriculture. In Namibia commercial livestock ranching constituted 80% of total agricultural output for 2000 (Institute for Security Studies Undated) with the farms covering 40% of the country. These farms harbor 90% of cheetahs (Morsbach 1986) resulting in human-wildlife conflicts (Joubert & Mostert 1975; Marker-Kraus *et al.* 1996; Marker *et al.* 2003a). According to Nowell (1996), about 9600 cheetahs were indiscriminately removed from the wild by live trapping or shooting in Namibia between 1978 and 1994. In most cases cheetah were caught or shot because they were perceived to attack livestock, but evidence that a cheetah was indeed the culprit was not always sought, and some removals were merely a preventive measure (Marker *et al.* 2003a).

A ramification of increased human population and agriculture is habitat fragmentation and animal or species isolation that, in turn, makes species such as the cheetah more susceptible to problems associated with small populations and disease outbreaks. Isolation plus low cheetah densities (Myers 1986) may induce high inbreeding rates with reproductive and physical abnormalities and may also decrease immune system resilience in the long-term (O'Brien *et al.* 1985). Crosier, Marker, Howard, Pukazhenthi, Henghali, and Wildt (*in press*) and Wildt, Brown, Bush, Barone, Cooper, Grisham and Howard (1993) revealed that up to 81% and 75%, respectively, of sperm produced by male cheetahs is pleiomorphic, which is mostly related to the lack of genetic diversity in this species. Likewise, formation of fragmented populations eliminates the possibility of new blood lines being introduced which increases the species' proneness to extinction. This is especially important when considering cases of disease outbreaks, a factor compounded by cheetah genetic uniformity (Wildt, O'Brien, Howard, Caro, Roelke, Brown & Bush 1987). This was the case for seven radio-collared cheetahs reported dead in the Etosha National Park after they had consumed a springbok infected with anthrax (Nowell 1996).

However, the impact of all these factors on the abundance of wild populations remains unknown (Munson, Fowler & Saunders 1993; Caro 1994; Stander 2004). For example, Laurenson (1994) provided little evidence that poor sperm quality has had a significant impact on the cheetah wild population in the Serengeti. Likewise, Crosier *et al.* (*in press*) when comparing the sperm quality between wild-caught versus captive cheetahs in Namibia, concluded that they produced comparable sperm of overall poor morphological quality. If this is combined with Howard *et al.* (1997) cited in Crosier *et al.* (*in press*), findings that a female can be impregnated with as few as $3.4\text{x}10^6$ total motile sperm using artificial insemination, questions arise regarding the direct impact or significance of the poor sperm quality on cheetah reproduction and subsequently on the species survival.

A third cause for concern over the survival of cheetah in Namibia is bush encroachment, the conversion of grassland into a vegetation type comprising mostly thorny bushes (e.g. *Acacia melifera*) (Quan, Barton & Conroy 1994; Bester 1996). Bester (1996) indicated that between 12 and 14% of land in Namibia is effected by this phenomenon. Whether this benefits or creates problems for wildlife, and in particular cheetah, remains contentious.

Purchase and du Toit (2000), McVitte (1979) and Purchase (1998) in Broomhall, Mills and du Toit (2003) argued that bush encroachment can be beneficial for cheetah as it provides (1) concealment against other predators or farmers (McVitte 1979; Mills, Broomhall & du Toit 2004), (2) ambush sites for prey, and (3) resting sites during the hot hours of the day. The critical factor is the level of encroachment. For example, Bailey (1993) attributed the decline of cheetah in Kruger National Park (KNP) to bush encroachment and Marker (2002) suggested that encroachment leads to injuries of the eyes or paws. Cheetahs then resort to predation of easy prey such as livestock, increasing human-wildlife conflict. Furthermore, Muroua, Marker, Nghikembua, and Jeo (*submitted*) suggested that bush encroachment is possibly altering the ungulate density and distribution on Namibian farmlands, thereby affecting the cheetah prey base. However, as for the previous threats, empirical evidence is unavailable.

The final threat to cheetah populations to be considered is the current protective legislation formulation and a lack of effective policing mechanisms in most African countries (Myers 1986; Hunter 2001). Although protected, cheetah can be destroyed if perceived to pose a threat to human life or property (Nowell 1996; Hunter & Balme 2004). Therefore, due to the lack of a requirement to provide evidence, carnivores in general still face persecution.

In summary, the extinction of cheetah could occur as a result of a combination of extrinsic and genetic factors that act independently (Mills 1996).

The challenges faced by cheetah have an impact on its current distribution within range countries, and this needs to be considered in the design of any methods for measuring populations. It is within this ecological framework and the utility distribution theory (Powell 2000), which refers to the intensity of use of an area by an animal, that cheetah habitat and use, prey and seasonality, gender and/or age, and reproductive status and home range aspects are reviewed here.

3.5 Cheetah habitat and its use

Although highly associated with open plains (Hamilton 1986), cheetahs utilize a variety and combination of habitats besides grasslands, such as open and dense woodland (Eaton 1974; Ray *et al.* 2005). In this thesis Lincoln, Boxshall and Clark's (1983) definition of "habitat" as being a particular type of environment occupied by an organism, is adopted. A study on cheetah habitat use by Broomhall *et al.* (2003) in the KNP, revealed that male cheetahs preferred more open habitats than the dense and thick woodlands. In contrast, females showed greater use of and preference for denser vegetation, and because such areas are associated with high prey densities, encounter rates are maximized for females. Marker (2002) reported similar results in the vastly bush encroached Namibian highlands, where females showed a preference for sparse bush nodes with high prey biomass. The preference for open habitats for males suggests that males should utilize roads often but this should be true to a lesser extent for females. This was found to be true for male cheetah in the KNP (Broomhall *et al.* 2003). Furthermore, being an open plains species, it can be speculated that preference for roads by both genders is prevalent when cheetahs are faced by exceptionally thick vegetation or bush encroached land.

Territory is defined as a protected part of an animal's home range (Burt 1943) that exceeds an equal-use pattern (Samuel, Pierce & Garton 1985) and that is more or less exclusive to an animal or group of animals of the same species (Lincoln, Boxshall & Clark 1983). Territoriality of any species requires demarcation of the territory. Territoriality in cheetahs is achieved through scent marking of conspicuous features, like solitary large trees (Caro 1994) and shrubs (Eaton 1974), or rocks and termite mounds (Caro 1994). In the KNP, landmarks close to roads were preferred (Broomhall *et al.* 2003). Cheetah scent marks simply deter intruders, which usually divert to a different route upon odor inspection (Eaton 1974). Females are reported to scent mark less frequently than males as such marking is primarily a communicative reproductive behavioural mechanism (Eaton 1974; Bothma & Walker 1999; Hunter & Hamman 2003).

3.6 Prey and seasonality

Cheetah have diverse prey with varying weights. Eaton (1974), quoting Graham (1966), Pienaar (1969), and Schaller (1972), indicated 27 different prey species for East Africa, 24 in the KNP and nine in the Serengeti. More recently, Hayward, Hofmeyr, O'Brien and Kerley (2006) using metadata analysis on 21 published and two unpublished studies from six different cheetah range countries, demonstrated that cheetahs prefer 15 prey species of which blesbok (*Damaliscus dorcas phillipsi*), impala (*Aepyceros melampus*), springbok (*Antidorcas marsulpialis*) and Thomson's (*Gazella thomsoni*) and Grant's gazelles (*Gazella granti*) are highly preferred over other species. Additional prey species reported by studies include the young of other antelope species, livestock such as sheep and goats (Nowell 1996), birds and, in high frequencies, hares (Caro 1994; Marker, Muntifering, Dickman, Mills & Macdonald 2003b; Wachter, Jauerning & Breitenmoser 2006). Preference for prey weighing less than 40kg is evident (McVittie 1979; Caro 1994); although Hayward *et al.* (2006) provides a higher maximum of 56 kg, with a minimum value of 23 kg and 36 kg as the optimum weight. Preying on species within this body mass range means that there is a shorter inter-hunting interval in order to meet the daily dietary requirement of 3 to 4 kg (Bothma & Walker 1999) or 2.8 kg (Frame 1999). Nowell (1996) estimated the prey capture interval to be two to three days in the Etosha National Park (ENP), Namibia. However, regarding prey size, both McVittie (1979) and Marker *et al.* (2003b) provided evidence that in the absence of competitors, cheetahs have a tendency to capture larger game.

Much cheetah prey is not sedentary but migrates with season to greener pastures subsequently affecting the daily movement and home ranges of cheetahs. Durant, Caro, Colins, Alawi, and FitzGibbon (1988) indicated that cheetahs migrate according to Thompson gazelle migration, with few individuals remaining in areas of low prey density. This was suggested to be a coexistence strategy of cheetah in competitive environments with lion and hyena (Durant 1998). As a result, spoor distribution and frequency can be considered to reflect variation in prey availability during particular seasons, which affects daily movements and home range determination. However, in Namibia the development of waterholes and fence erections reduced and to a certain extent stopped the migration of game which is consequently more resident on farmlands (Marker 2005 *pers. comm.*).

3.7 Gender, age and reproductive status and effects on movement

All animal movements are influenced by gender, sex ratio, age, and reproductive status. Marker (2002) and Caro (1994) found that both juveniles and prime adult male cheetahs live a nomadic life prior to acquiring a home range, if one is ever acquired. This means that prior to this they are required to travel long distances, until an abandoned or a non-occupied suitable habitat is encountered. Purchase and du Toit (2000) presented similar results for a re-introduced population of cheetahs (n=3 males and 1 female with cubs) in the Matusadona National Park, Zimbabwe (MNP). In that study, cheetahs initially exhibited wider movements and home ranges, but after establishing a territory, they exhibited a reduction in movement and constriction in range.

Females cheetah, however, are reported to travel further on a daily basis than males but this is altered with their reproductive status. Literature indicates that unaccompanied females traverse larger areas than those lactating or with cubs older than four months (Laurenson 1994; Durant 1998; Kelly, Laurenson, FitzGibbon, Collins, Durant, Frame, Bertram & Caro 1998; Marker 2002). A number of reasons have been postulated for female movements including, (1) searching for prey (Gittleman & Harvey 1982; Stern 1998), (2) locating effective and secretive lair sites (Laurenson 1994; Kelly *et al.* 1998), (3) avoiding other carnivores (Durant 1998; Mills *et al.* 2004) and (4) familiarization of habitat by mothers to young cubs (Marker 2002).

3.8 Home range

Although criticized for vagueness (White & Garrot 1990; Powell 2000), Burt's (1943) definition of home range, "an area traversed by the individual in its normal activities of food gathering, mating, and caring for young", remains the most widely accepted. Morsbach (1986) estimated a mean home range of 932, 1522 and 651 km^2, respectively, for an adult male, female and pair of adult cheetahs on Namibian farmlands. Marker (2002) reported similar results (n=23) for annual range variations using the minimum convex polygon (MCP's) method (overall average 1056km^2 $\pm$ 791km^2) with males utilizing smaller ranges. Furthermore, lifetime ranges averaged 1642 $\pm$ 1565 km^2 for 28 cheetahs studied. Bowland (1994) used photography to calculate smaller home ranges in the KNP, of 500, 770, and 193 for male, female and coalitions of three, respectively. Caro (1994), using the same methodology, reported even smaller home ranges in the Serengeti of 37.4, 777.2 and 833 km^2, for a coalition, a non-territorial single male and a female respectively. From these studies, a trend is evident: females have larger home ranges than single territorial males, who in turn have smaller ranges than nomadic males but larger ranges compared with coalition males.

A concept related to home range, and already alluded to in Section 3.5 is that of an animal territory or core area (Burt 1943; Samuel *et al.* 1985; Powell 2000). Although, there is debate about exactly what comprises an animal core area and how it should be determined, both Burt (1943) and Powell (2000) suggested that this could be an animal home range in its entirety or only part of it (e.g. areas with high expenditure of time by animals). This has implications for spoor distribution, and on the planning of studies intending to use spoor frequency as an index of abundance.

3.9 Conclusion

The ecological background of a species must be considered and understood for the development of any abundance index, including spoor frequency. Human activities and attitudes do, however, also influence distribution both inside and outside any study area for population census, and this also needs to be considered.

Chapter 4

MONITORING CARNIVORE ABUNDANCE: THE USE OF CENSUS TECHNIQUES

4.1 Introduction

Monitoring programs are concerned with the detection of changes over time or the measurement of trends (West, McDaniel, Smith, Tueller & Leonard (1994) in Watson & Novelly 1994) at the four-biodiversity levels, i.e. the landscape, ecosystem, species or population and genetic levels (Gaines, Harrod & Lehmkuhl 1999). The present study focuses on the third level. A monitoring system comprises several stages (see Watson & Novelly 1994). The first is the continuous standardized sampling of a species attribute, for example spoor, at fixed locations over extended periods of time. This is followed by an analysis and interpretation stage, and lastly by the incorporation of the new information into existing monitoring schemes, thus enhancing these. However, the incorporation of new information into monitoring programs is not always conducted (Karanth, Nichols, Seidensticker, Dinerstein, Smith, McDougal, Johnsingh, Chundawat, & Thapar 2003). Furthermore, mostly due to financial constraints, estimates which are inferences based on samples of a wider population, are often used rather than total counts (true abundance).

A number of census techniques have been employed to estimate relative abundance for carnivores (methodologies reviewed in Caughley 1977; Seber 1982; Lancia, Nichols & Pollock 1994; Wilson, Cole, Nichols, Rudran & Foster 1996; Caughley & Sinclair 1997; Mills 1997; Mahon, Banks & Dickman 1998; Gese 2001; Wilson & Delahay 2001; Sadlier, Webbon, Barker & Harris 2004; Kunkel, Mack & Melquist 2005; Gese *Undated*). Relative abundance is defined as the total number of animals per unit area on one sampling occasion in relation to others separated in time or space (Wilson & Delahay 2001). Census techniques to estimate densities are categorized as either direct or indirect (Lancia *et al.* 1994).

4.2 Direct methods

Direct methods are further subdivided into total and partial count methods (Lancia *et al.* 1994) but both involve a direct count of animals. The mark-recapture method has been employed with capture of lions in the Kgalagadi Transfrontier Park (KTP) (Castley, Knight, Mills & Thouless 2002), and through sightings with photography of tigers (*Panthera tigris*) (Karanth 1995) and of cheetahs (Bowland 1994; Kelly *et al.* 1998). In addition, Caro (1999) employed line transects to estimate abundance of lion and spotted hyenas in woodland vegetation with visibility as the major covariate, and Nowell (1996) described a reporting system of kills or sightings for cheetah in Namibia. Radio telemetry has also been used to derive cheetah density estimates (Gros, Kelly & Caro 1996; Marker 2002) but mostly this is used for home range determination (Caro 1994; Purchase & du Toit 2000; Marker 2002; Broomhall *et al.* 2003) from which density can be derived. Rabinowitz (1997) however, classified radio telemetry as an indirect method of density estimation.

4.3 Indirect methods

Indirect methods are also referred to as population indices, which are counts that are assumed to be related to true density but which are not expressed in terms of density (Eberhardt 1978), and these rely on the presence and detectability of a sign of an animal's presence (Wilson *et al.* 1996). The presence of tracks along transects has been widely investigated for different species. Dzieciolowski (1976), van Dyke, Brocke and Shaw (1986), and Smallwood and Fitzhugh (1995) used this method on mountain lion (*Felis concolor*), Funston, Herrmann, Babupi, Kruiper, Kruiper, Jaggers, Masule and Kruiper (2001) employed this method for lions, whereas Stander (1998) used it for leopard and wild dogs, and Karanth *et al.* (2003) for tigers. Gusset and Burgener (2005) also used track counts and measurements to estimate abundance of leopard, caracal (*Caracal caracal*), serval (*Leptailurus serval*), African wildcat (*Felis silvestris libyca*), brown hyena and black-backed jackal (*Canis mesomelas*). These studies made reference to the appropriateness of this method for other carnivores like cheetah and wild dog as well for ungulates. Scent stations with counting of tracks has been used for feral cats (*Felis catus*), red foxes (*Vulpes vulpes*) and dingoes (*Canis lupus dingo*) (Mahon, Banks & Dickman

1998) and for jaguar (*Panthera onca*), jaguarandi (*Felis yagouarundi*), ocelot (*Felis pardalis*) and puma (*Puma concolor*) by Harrison (1997). Other examples of indices include auditory indices used by Ogutu and Dublin (1998) for lions and spotted hyenas; pellet counts or molecular scatology for red foxes (Cavallini 1994; Webbon, Baker & Harris 2004); physical structure surveys for black-tailed prairie dogs (*Cynomys ludovivianus*) (Mallick, Driessen & Hocking 1997; Severson & Plumb 1998); and interviews and questionnaires for cheetahs (Joubert & Mobert 1975; Myers 1975; Gros *et al.* 1996; Marker-Kraus *et al.* 1996; Gros 1998; Gros & Rejmanek 1999; Orford 2002).

4.3.1 Use of indirect methods as indices of abundance

Indirect methods can be used in three ways as indices of abundance (Eberhardt & Simmons 1987): firstly by utilizing the index count as an estimate of abundance without its evaluation against a "more reliable" density estimate, which means that it is derived independently, or secondly, it can be used by evaluating it against a "more reliable" density estimate usually through a linear regression. Thirdly, an index of abundance can be used after being modified by a linear equation derived from comparisons with another method of measuring density. Skalski, Ryding & Millspaugh (2005) suggest a fourth way of using indices, which is to compare relative abundance of a species across time or space. The present study relates to the second and fourth points as it attempts to compare spoor density and spoor frequency indices with population estimates obtained through radio telemetry and to examine changes in spoor density and frequency over time relative to the population changes estimated through radio telemetry.

4.3.1.1 Analytical framework for indices

An ideal index is a constant-proportion type index (Overton 1980), which was simplified by Eberhardt and Simmons (1987) as $E(n) = p N$, where $E(n)$ is the expected index value, p is the observality constant and N the true population size. By isolating N one obtains an estimate for the population in the study area, $N' = C / \alpha p$, where N'= estimated population size in the study area, $C = E(n)$, α is the proportion of total area actuallysurveyed, and p remains as the coefficient variable relating N' and C (Lancia *et al.* 1994). The key parameter in both equations is the constancy value and how this is estimated (Overton 1980). If the assumption that ($p=\alpha$) =1, was always valid, then an index could be considered to be a total census. However, this is not always the case, and p can be postulated to be <1. Caution must be taken during analysis, in that absence of a sign may indicate presence but not at that particular sampling plot (Caughley 1977).

Other underlying assumptions applicable to all other census tactics are that all signs are counted and animals display no tendency of clustering. Double sampling, where two different sampling methods are used, according to Eberhardt and Simmons (1987), accommodates these variations. Calibration is therefore vital prior to the use of an index as a means to estimate density.

4.3.1.2 Index calibration and its requirements

Calibration of an index can be defined either as the adjustment of an index to account for the recognized effect of some extraneous variable (Overton 1980), (e.g. season, animal behaviour, human error and weather); as the estimation of the relationship between indices and abundance (Eberhardt & Simmons 1987) or else as the evaluation of an index against a presumed more accurate/reliable density estimate of abundance (Skalski *et al.* 2005). A model of calibration, such as a linear equation, is "a test of a model with known input and output information that is used to adjust or estimate factors for which data are not available" (Ford 2000). Thus, a calibration equation is achieved by either linear or multiple regressing of a true density on an index (Vincent, Gaillard & Bideau 1991; Brown, Moller, Innes & Alterio 1996; Severson & Plumb 1998; Stander 1998; Walker *et. al.* 2000; Funston *et al.* 2001; Watson 2003).

Calibration does not automatically convert an index into a measure of population density (Overton 1980) but permits (1) the testing of the constancy assumption between the index and a true density estimate determined independently (Conroy 1996), and (2) improves overall precision (Overton 1980) plus sensitivity, and lastly, if (1) and (2) are attained allows for its conversion into an estimate of abundance (Eberhardt & Simmons 1987). It is through calibration that p is derived, being equivalent to the slope of a regression line (Conroy 1996). Therefore, a reference density estimate obtained independently from the index, but within the same time frame or with only a slight time difference, within the study area, is necessary. This is defined as double sampling (Lancia *et al.* 1994).

4.3.1.3 Double sampling and indices precision

Double sampling although costly (Stander 1998), provides an overall cost-effectiveness (Eberhardt & Simmons 1987; Pollock *et al.* 2002), as it allows for the sampling of a desired variable, i.e., spoor, at large scales and for longer time periods using a less expensive method while a more accurate method, such as radio telemetry or mark-recapture (physical or via camera trapping), is employed for shorter time periods and smaller sampling areas. This allows for the estimation of a correction factor. Vincent *et al.* (1991) calibrated the kilometric index, defined as the number of roe deer (*Capreolus capreolus*) observed by the number of kilometers traversed, against an actual population size determined using the mark-recapture technique. Brown *et al.* (1996) and Severson and Plumb (1998), also calibrated density estimates obtained by using prairie dog burrows with those estimated via mark-recapture and visual observations. The three evaluations indicated that these indices were not suitable for estimating density.

Through bootstrapping analysis, 95% confidence intervals (CI) for calibration coefficients can be determined as a precision measure. For example, Watson (2003) calibrated total counts using mark-recapture density estimates, for the New Zealand fur seal pups (*Arctocephalus forsteri*), with the intent of establishing whether total counts were a good index of abundance for fur seal pups. Her calibration model 95%CI was narrow: 1.33(TC) + 0.293(#CR) - 14.9 and 1.07(TC) +0.198(#CR)-36.13 (where TC= total counts and #CR number of crevices in fur seal colonies). Walker *et al.* (2000) also obtained similar results, 10.6 (8.9 – 12.3), when they calibrated faecal pellet counts against visual counts for the mountain vizcachas (*Lagidium viscacia*).

During calibration and double sampling studies it is assumed that ground counts, such as spoor tracking, are exempted from error. However, as Caughley (1977) points out, both spoor counts and radio telemetry are random, prone to variation and human measuring error and subsequently the estimated slope of a calibration model is biased downward. Thus, calibration equations may underestimate abundance. A further requirement for the use of an index, in addition to double sampling and calibration, is the requirement of a relationship between an index and true abundance estimate.

4.3.1.4 The relationship between index and true abundance

The association between an index and true abundance, however, is not always linear, as the distribution of animals is rarely normal. Caughley (1977) pointed out that frequency indices tend to display non-linearity more often than linearity, particularly for populations of low density (Edwards, de Peru, Shakeshaft & Crealy 2000), which is the case for cheetah (Caro 1994). Greenwood (1996) stated that highly variable samples tend to have lower mean indices for a particular mean population size and such samples are usually indicated by a convex curve on an effort precision graph. The converse also holds true, with less variable samples having more precise mean index estimates (indicated by a concave curve). Nevertheless, in such instances of high variability data transformation, i.e., log or arcsine (Eberhardt 1978; Krebs 1989) allows for counts to approximate normality while accounting for external factors that may affect an index (Eberhardt 1978).

In general, an index is related to true abundance by a coefficient (Nichols & Karanth 2002) or a ratio acquired through double sampling (Lancia *et al.* 1994). For an index to be considered a reliable estimate of abundance, it must depict an association with a true density estimate that is positive, linear and monotonic (Conroy 1996), with a zero intercept (Lancia *et al.* 1994) and a slope of one (Poole, Cowan & Smith 2003). Furthermore, an index also needs to be sensitive, able to detect changes or differences (Anderson 2001 but see Anderson 2003 and Engeman 2003). This ratio can fluctuate with density or exogenous variables such as weather conditions, and it is precisely due to this that a need for calibration exists, to establish with a certain degree of confidence whether population abundance for a species can be predicted by the index.

Having reviewed what constitutes a direct and indirect census method, and how abundance and densities are derived for monitoring, the next section presents a review of some of the most common currently employed census methods.

4.4 Previous methodologies and methods employed to census cheetah

Five census methods are reviewed in this section: questionnaires and interviews, photography and mere encounters, spoor tracking, mark-recapture and radio telemetry. The emphasis is on aspects of practicality and feasibility, repeatability (for variance estimation), and the accuracy of derived estimates.

4.4.1 Questionnaires and interviews

The use of questionnaires and interviews is by far the most commonly employed tactic to estimate cheetah population density (Joubert & Mobert 1975; Myers 1975; Gros *et al.* 1996; Marker-Kraus *et al.* 1996; Gros 1998; Gros & Rejmanek 1999). These methods have been widely used because they have a much greater audience in terms of individuals and geography and were considered to be relatively inexpensive by Mills (1997) although this benefit was disputed by Clarke (1986). Simultaneously, ancillary information regarding management and conflict aspects can also be collected (Marker-Kraus *et al.* 1996). These methods are, however, time consuming, require thorough preparation and supervision (Clarke 1986), responses can be of poor quality, inadequate and limited (Mills 1997), and results are prone to reporting error in addition to being dependent on sociological variables such human attitude towards an animal (Cavallini 1994), all of which make spatial comparisons difficult. Nowell (1996) reported this to be a major problem encountered in Namibian surveys where farmers reported having up to 100 cheetahs per farm, which could either be due to high cheetah mobility and thus over-counting or else due to perceptions. Bashir, Daly, Durant, Forster, Grisham, Marker, Wilson and Friedmann (2003) also identified the lack of accuracy determination associated with this method as another weakness. Nevertheless, when Gros *et al.* (1996) regressed interviews against cheetah prey biomass, and cheetah density averages across 13 protected areas in East Africa, they identified interviews as the most accurate method of estimating cheetah density.

4.4.2 Photography and mere encounters

Opportunist photography by staff and tourists (Bowland 1994) or scientists (Kelly *et al.* 1998; Kelly 2001) and mere encounters (Caro 1994), rely either on recognition of individual animals based on their unique coat patterns or else on the identification and ascription of imprints to individuals (Smallwood & Fitzhugh 1995; Stander 1998; Wilson & Delahay 2001). However, Funston *et al.* (2001) suggested that it may not be necessary to link individual identity and spoor imprint in lions.

Opportunistic photography and mere encounters can be considered to be cost-effective and easily implementable, allowing equally for demographic and environmental pattern investigation, and for home range determinations (Bowland 1994). Problems associated with this method, however, are that long sampling periods are required, which means that it is not practical for immediate use, and if remote camera stations are deployed (e.g. Karanth 1995; Karanth & Nichols 1998; Wallace, Gomez, Ayala & Espinoza 2003; Silver, Ostro, Marsh, Maffei, Noss, Kelly, Wallace, Gomez & Ayala 2004): (1) they are prone to be stolen and/or damaged by other animals, (2) the researcher may incur high film costs due to non-target species (Conroy 1996) although using digital photography could relatively reduce operating costs against higher investing costs and (3) appropriate locations for their placement may not be present or be rare within a study area (Kelly 2001). An assumption of this method is that no animal within the sample area has a zero probability of being captured (Silver *et al.* 2004). The use of play trees as key sites for deployment of cameras has been proposed to alleviate the latter problem (Bowland 1994; Nichols & Karanth 2002; Wallace *et al.* 2003), increasing the probability of captures. Karanth and Nichols (1998) used the same approach for tigers. They recognized, however, that the study site boundaries are not defined by the location of traps, and it is necessary to determine a boundary width area (Otis *et al.* 1978; Krebs 1999).

4.4.3 Animal imprint frequency

The third index considered is animal imprint frequency or spoor frequency. There are two methods employed for collecting spoor as a means to monitor population trends (Nichols & Conroy 1996; Wilson & Delahay 2001); (1) counts along transects and (2) counts at scent stations, with only the former being considered here. This method requires that transects are subdivided into units, which can be either kilometers or meters, to allow for unit comparisons over time and space (Vicent *et al.* 1991; Wilson & Delahay 2001). Transect placement need not be random if the aim is to estimate relative abundance, but random placement of transects is necessary for absolute estimates (Wilson & Delahay 2001). When multiple transects are employed spatial independency should be sought to minimize serial correlations (Edwards *et al.* 2000; Wilson & Delahay 2001). According to Funston (2005 *pers. comm.)* transect lengths should be at least as long as the average home range size of an animal within the study area. Thus, prior knowledge of target species' ecology and movement patterns is vital during transect design for abundance estimates. Similarly counts should be independent to allow for the normal distribution of individuals. As a result an index is a statistic (the total number of spoor counted), presented (1) as a ratio, the statistic over a unit of space or time (Davis & Winstead 1980), (2) a frequency index (Dice 1948; Lancia *et al.* 1994), with the proportion of sample units that recorded or did not record an animal sign or (3) as a count per area of search (Henderson 2003).

Spoor tracking, including linking spoor to individuals, has been reported to be cost-effective (Smallwood & Fitzhugh 1995), objective and repeatable (Caughley 1977; Martin & de Meulenaer 1988), non-invasive and as for photography and mere encounters, provides additional information including demographics (Liebenberg 1990; Stander 1998; Alibhai 2004). In other carnivores such as lions and leopards, spoor density is reported to be a function of true density (Smallwood & Fitzhugh 1995; Stander 1998; Funston *et al.* 2001). Liebenberg (1990), Stander (1997) and Alibhai (2004) highlighted the importance of employing indigenous people to improve track identification and ascription of individual spoor to individuals, when this is necessary. Spoor tracking can also be incorporated into existing conservation and other management practices.

However, as expected for large, mobile organisms, it requires large sample sizes to attain reliable estimates (Lettink & Armstrong 2003; Kunkel *et al.* 2005) due to high variances at low sampling levels (Mooty & Karns 1984 cited in Servin, Rau & Delibes 1987; Stander1997; Funston *et al.* 2001). Stander (1998) suggested a sampling effort of 1200 km and Funston *et al.* (2001) a slightly lower value of 1100 km for variance stabilization, for leopards and lions, respectively.

Factors that may influence spoor tracking as a method of measuring carnivore abundance and density include: (1) substrate suitability, i.e. soft and dusty roads, which according to Cavallini (1994), Mallick *et al.* (1997) and Sadlier *et al.* (2004) may restrict the use of this method to certain seasons; (2) environmental heterogeneity, topography or vegetation as they impact on the random deployment of signs; (3) animals' movement patterns (Stern 1998) that in turn affect (4) sign detectability, and (5) decaying factors (weather, time of day, habitat, number of other animals) (Wilson *et al.* 1996; Stern 1998). Of the above factors, the interaction between substrate and detection probability is pivotal (Funston 2005 *pers. comm.).* Furthermore, the assumption is that the position of transects within the study area does not alter normal animal behaviour (i.e. repel or attract them) (Servin *et al.* 1987). Dzieciolowski (1976) referred to a study by Brunnett and Lamdon (1962), who investigated the relationship between the number of tracks and animals crossing a defined transect within enclosures. They concluded that track counts detected population changes but not their magnitude. In these studies however, there was a lack of validation of this and other indices against direct estimates to interrogate their effectiveness as monitoring methods (Eberhardt & Simmons 1987; Stander 1998; Walker *et al.* 2000). There is evidence of a linear relationship between spoor and true density for leopards, wild dogs and lions (Stander 1998; Funston *et al.* 2001), hence the need for calibration of this method for free-ranging cheetahs, as proposed by this study.

4.4.4 Mark-recapture

Mark-recapture (MR) as a census technique has also been employed to estimate abundance of carnivores. Initially, it required the physical capturing, marking (e.g. ear tags, radio-collars) and re-capturing of individuals, so that capture history tables for individuals could be developed and consequently abundance estimates (plus densities) determined for a study area. The physical capture and marking is no longer a requirement for implementing this method provided that a mechanism to identify individuals per sampling occasion is feasible. For example Karanth (1995) and Karanth & Nichols (1998) employed the MR framework using photography on tigers, as they can be distinguished by their coat patterns. Likewise, Wallace *et al.* (2003) and Silver *et al.* (2004) also employed photography capture trapping in the Amazon, Belize and Bolivia for jaguars (*Panthera onca*). Kelly (2001) recommended the use of automated photo identification programs to aid with the animal identification. She found this to be robust even to matcher inexperience.

Abundance estimation using the MR methodology is dependent on the objectives of the study and its design. For absolute abundance, closed population models are necessary, that is, the study should be conducted in a shorter time interval to ensure that the geographic and demographic closure assumptions are upheld (Lettink & Armstrong 2003). For most long-term monitoring programmes, open population models such as the Jolly Seber method (Krebs 1999) should be applied as these assumptions are usually violated. Therefore, open models should be interpreted more as a relative abundance estimator. Nichols (1992), and Nichols and Karanth (2002), however, suggested that for long-term monitoring programmes both closed and open models should be applied, particularly as available analytical programs (e.g. CAPTURE, MARK) for open models are less robust to capture heterogeneity. Capture heterogeneity refers to either a response to a trap (animal shyness or happiness), or trap placement within an animal home range/territory or sampling effort and animal age dispersal (Lettink & Armstrong 2003) which may have either a positive or negative impact and bias abundance estimates. MARK models these sources of bias in addition to testing their goodness-of-fit (Cooch & White 2005). The latter, in conjunction with the Akaike Information Criterion, are used for final model selection (Cooch & White 2005).

Other assumptions for closed models include: (1) marks are not lost or overlooked and (2) all animals have the same probability of being captured throughout the study. For open models, in addition to these two assumptions, it is assumed that all animals of the same type have the same survivorship probability and that sampling time is minor in relation to intervals between samples (Krebs 1999; Lettink & Armstrong 2003).

Most carnivore conservation organizations in Namibia, specifically the Cheetah Conservation Fund, Africat and the Okatumba Wildlife Research (OWR), do ear tag handled cheetahs (Forster & Forster 2003; Marker *et al.* 2003a; Conradie 2005). Besides this, the OWR at a certain stage did intentionally set traps in an attempt to capture and mark every cheetah within their study area (Forster & Forster 2003). The reasoning behind the marking of animals was to eventually apply MR models and derive national abundance estimates. Ancillary information on the species' ecology (habitat distribution and utilization), demography, genetics and morphometry are also collected during handling sessions.

The limited use of the mark-recapture technique is related to the difficulties involved in meeting the underlying assumptions as well as the logistics. The two crucial assumptions as mentioned are the demographic and geographic closure (Mills 1997; Lettink & Armstrong 2003). For a study to uphold such assumptions shorter time intervals between capture and recapture are necessary (Lettink & Armstrong 2003), but at higher sampling effort and intensity. However, autocorrelation, the process whereby animals are not given sufficient time to resume natural patterns (Stern 1998) may affect estimates thus the selection of time intervals during a project design stage require consideration. Even if the assumptions are met, low capture and recapture rates may demand a longer sampling period (Sadlier *et al.* 2004) so that sufficient sample sizes are acquired, at the cost of resorting to open population models for abundance determination. Furthermore, acquiring a reasonable statistical sample size may mean that financial costs can be substantial, and this is also influenced by the recapture methods. Thus, with or without physical capture MC can be a highly costly and labour demanding method (Lettink & Armstrong 2003; Sadlier *et al.* 2004). Lastly, the MR method tends to provide only an estimate of true abundance, as the sampling units are a subset of a wider unknown zone (Lancia *et al.* 1994; Sutherland 1996; Karanth & Nichols 1998) unless the total area is predetermined using the mean maximum distance moved method as described by Karanth and Nichols (1998), Krebs (1999) and Jackson, Roe, Wangchuk & Hunter (2005) or through radio-telemetry.

Therefore, for MR the number of capture sessions, timing and trapping strategy are the major factors for consideration, prior to undertaking any MR study (Lettink & Armstrong 2003).

4.4.5 Radio telemetry

Radio telemetry (RT) focuses on the distribution of animals (Stern 1998), their movements and migration, and provides accurate information on home range use and size, and social organization (Karanth 1995). From this information population densities can also be estimated (Morbasch 1986; Gros *et al.* 1996; Stander 1998; Marker 2002) although this is done only infrequently (Minta & Mangel 1989).

Radio telemetry requires capturing of individuals, selection of appropriate collars and fitting them, insurance that collars have no effect on animal behaviour, and release (White & Garrott 1990). Tracking can either be on foot, by car or airplane depending on the resources available which also determines the tracking frequency. During tracking, radio signals (translated into location points) are transmitted continuously or at intervals (MacDonald 1978). Three major limitations of radio tracking are (1) the sample size of animals required to get an adequate statistical sample (White & Garrott 1990), (2) it provides no biological reason for why an animal may spend disproportional amounts of time within different micro habitats of its range (MacDonald 1978), and (3) the high cost of equipment.

The mostly commonly employed analytical method to ascertain home range size is minimum convex polygons (MCP) (Harris, Cresswell, Forde, Trewhella, Woolard & Wray 1990; Purchase & du Toit 2000; Marker 2002; Broomhall *et al.* 2003). MCP uses the outer fixes registered during an animal tracking period to delineate home range

(Harris *et al.* 1990; Powell 2000). However, this constitutes a weakness of the methodology because according to Harris *et al.* (1990) areas outside an animal home range such as excursion zones may be included thus inflating home range sizes. Secondly, MCP provide no indication of the intensity of range use, although these data may be available depending on sampling intervals (Samuel & Fuller 1994). Another analytical method employed to determine densities using radio tracking of locations is the kernel density estimator (Powell 2000). A kernel is a three-dimensional elevation, drawn for each radio location obtained from tracking an animal (Powell 2000). The density or clumping of these kernels, plus their band width, provides an indication of an animal's time length expenditure for different areas, allowing for the demarcation of contours according to such lengths (Powell 2000). Therefore, core home ranges can be determined more easily using the kernel method than the MCP.

Density estimates using radio telemetry are determined by estimating the number of collared or marked animals present in a study area at a particular time (Gros *et al.* 1996; Marker 2002) whether the animals are solitary or social (Gese *Undated*). The extraction of overlapping areas allows for the determination of an overall range for individuals in a particular area (Stander 1991) thus the necessity for precise fixes. The length of sampling periods should be long enough to allow an animal to describe its home range boundaries (Otis & White 1999). For solitary species, density can be determined using home range sizes, the level of inter and intra sexual overlap plus the percentage of radio collared floaters utilizing the area (Gese *Undated*). Home ranges are used instead of territory for density estimates as these are more representative of the area used by animals than territory, However, as Burt (1983) suggested, if a territory is an individual's entire home range, the territories could be used to estimate density.

Some of these problems can be ameliorated by employing discontinuous tracking and concave and restricted polygons (Harris *et al.* 1990). Thus, in order for radio telemetry to be a more efficient technique for density estimation a moderate level of skill is required (Sadlier *et al.* 2004). Nevertheless, radio tracking is a robust model and even with a low number of fixes, provides comparable information. For example, Nilsen, Herfindal and

Linnell (2005) compiled a database (totaling 199 studies) using published and unpublished telemetry data to study the intra and inter variation in home range size determined by RT for 12 carnivore species. RT also facilitates direct observation which allows for the collection of ancillary information such as prey abundance, mortality, survivorship, and reproduction (Sadlier *et al.* 2004).

The fact that a researcher can establish direct contact with animals, continuously or periodically, particularly for social animals such as cheetahs, makes RT a preferable census method provided that there is due attention paid to the limitations mentioned above.

4.5 Conclusion

The intent of this chapter was to provide an overview of the diversity of methods, direct and indirect, that have been utilised for censusing carnivores, including cheetahs, and to assess advantages and problems of the different methods. The selection of a census technique is a function of a number of factors including the study area size, aims of the study, available resources, the behavioral ecology of the targeted species (Wilson & Delahay 2001), as well as the suitability of other methodologies (Lettink & Armstrong 2003). Of these, the behavioral ecology of the study carnivore species can be considered to be the most important factor guiding the selection of a census technique.

Chapter 5

METHODOLOGY

5.1 Methodology

Chapter 2 provided the overall study framework. This section in turn, provides the link between this framework (wildlife monitoring) and the statistical framework components. It focuses on the objectives for the selected census methods (radio and spoor tracking), data collection and analysis techniques as well as on the interpretation and application of results in relation to the identified study objectives.

5.1.1 Objectives, data collection and analysis selection procedures

5.1.1.1 Radio-tracking

Research on the spatial distribution of cheetahs in north-central Namibia, the study area, was conducted by Marker (2002). Although, density estimation was not part of her primary objectives, she provided minimum annual cheetah density from 1993 to 2000. Density was determined by dividing the overall area covered by the collared animals by the number of cheetahs tracked per year, and this is expressed per $1000km^2$. A brief description of her radio tracking sampling protocol is presented here.

From 1995 to 2000 a total of 41 individual cheetahs were radio collared (with an annual average of 12.67 ± 2.88 collared cheetahs), with some being tracked for more than a year. Cheetahs were fitted with radio collars with an external antenna (Advance Telemetry Systems, Minnesota) which had a life expectancy of 36 months. These were initially tracked twice a week, between January 1995 and May 1996, but due to financial constraints as of May 1996 to December 2000 tracking was conducted only once a week. Tracking was done using a fixed-wing Cessna 172 plane. Attempts were made to locate every radio-collared animal per flight and once radio signal was detected a portable Global Positioning System (GPS) was used to record the geographical location. Attempts were also made to obtain a sighting of the animal during these flights.

Minimum convex polygon home range (95%) estimates were calculated using the GPS fixes. ArcView GIS (Spatial Movement Analysis extension Version 3.2, ESRI, Redlands, CA) was used to calculate annual home ranges for animals that were tracked for more than a year. Asymptotes for the number of radio collared fixes were determined to avoid density bias (i.e. underestimate) which can be introduced when using a small number of radio locations (Kunkel *et. al.* 2005).

5.1.1.2 Spoor tracking

Two designated farm roads, hereafter referred to as transects (A and B) (Figure 2), were used as spoor tracking sampling routes from 1995, on the farm Elandsvreugde, the CCF headquarters (16°39'0" E, 20°28'12" S). The farm falls within the radio telemetry study area. The transects were initially designed for strip counts to monitor game abundance and habitat preference of the different game species. The transects are adjacent to each other with zones of overlap and maximum distance of about 6km at furthest points approximately 26.64km and 28.54km in length. Determining the type of relationship between strip counts and spoor tracking for various game species emerged as an objective at a later stage. Both ungulates and carnivore spoor were collected during spoor tracking sampling sessions. Hasheela (2004) quantified this relationship as being positive for some ungulates and predators. However, a weak relationship occurred for cheetah due to no visual sightings. Reviewed literature suggested the possibility of using the same data for a different analytical framework (Vincent *et al.* 1991; Smallwood & Fitzhugh 1995; Stander 1998; Funston *et al.* 2001). Therefore, the same data were used to calibrate spoor tracking against radio telemetry for cheetah in this study.

Four objectives for the current study were identified: (1) to determine cheetah density per unit of effort (km) and area (km^2) based on spoor frequency, using the spoor tracking surveying method; (2) to investigate the relationship between cheetah densities measured from two indirect census techniques (spoor tracking and radio telemetry); (3) to determine the optimum sampling effort required to obtain reliable spoor frequency counts (determine sampling effort in kilometers required to minimise variability in counts); and (4) to determine the sensitivity of the spoor frequency census method to actual annual changes in cheetah density as measured by radio telemetry.

Spoor tracking was conducted on a monthly basis between 8h30 – 11h00, giving a total of 3482km for 63 samples for the five-year study period. A vehicle (either a Toyota Venture, Model 1995 or an Isuzu Model 1998) was used during tracking, traveling at constant speed of 10km/hour, with a tracker seated on the bonnet. The same tracker was used on most spoor tracking sampling occasions, for consistency and as means of reducing the amount of human error. Once a cheetah spoor was detected, the vehicle was halted, the spoor was verified and the following were recorded: the associated odometer reading or sample location (transects are demarcated with metal poles at 0.53 ± 0.02 on transect A and 0.60 ± 0.08 on transect B intervals), the direction in which the animal was heading, and a count of the spoor. Only the most recent spoor, i.e. assumed to be spoor from the night prior to tracking due to its clarity, and with a high degree of certainty of belonging to cheetah were recorded, as the intent was to derive an estimate of the current density (Rabinowitz 1997). Analysis procedures were identified per objective, as follows:

Objective 1: Index densities are usually expressed per unit of effort, for example spoor per kilometers or scat/kilometer/day (Smallwood & Fitzhugh 1995; Stander 1998; Funston *et al.* 2001; Kunkel *et al.* 2005; Gese *Undated*). Therefore, the first objective for this study was determined by dividing the cumulative number of fresh cheetah spoor by the total kilometers covered per annum (Stander 1998; Funston *et al.* 2001). Each transect had its annual cheetah density calculated in this manner. Data from both transects were also pooled together to increase the index power (after Funston *et al.* 2001). It is anticipated that serial correlation could have occurred due to transect proximities. Thus, individual transect densities are likely to be similar.

Objective 2: The type of relationship existing between an index and a true density estimate is the factor determining whether an index can indeed be confidently used as a monitoring technique. The least-linear regression model analysis was identified as the most appropriate method to establish this relationship (Durrheim 2002), and this selection was vastly corroborated by reviewed literature (van Dyke, Brocke & Shaw 1986; Vicent *et al.* 1991; Smallwood & Fitzhugh 1995; Stander 1998; Funston *et al.* 2001; Poole *et al.* 2003; Watson 2003). The strength of the relationship was ascertained using correlation coefficients (Durrheim 2002).

Objective 3: The determination of optimum sampling effort required to minimize sample variability is vital for two reasons: (1) financial resource constraints and (2) determination of sampling time series (i.e. when the next stage of a monitoring process begins). Thus, plotting spoor frequency (the number of kilometers per spoor) or the coefficient of variance (standard deviation divided by the sample mean of the counts) against cumulative sampling effort in kilometers (Greig-Smith 1983; Stander 1998; Gibbs 2000; Funston *et al.* 2001) provides an indication of the level whereby further sampling presents little gain in terms of precision. A large number of samples are usually required for indices, particularly for species that live at low densities and are wide ranging, as these tend to render a high number of samples with zero spoor frequency. Cheetahs have been described by Caro (1994), and Hunter and Balme (2004) to live at low densities and to be wide ranging. For these reasons, higher sampling efforts for spoor frequency or coefficient of variance to reach an asymptote are expected for cheetah in comparison to sampling effort asymptotes values from other studies: 1200km for leopards (Stander 1998) and 1100km for lions (Funston *et al.* 2001). The determination of asymptote levels will be subjective (Greig-Smith 1983) and guided by the decrease in the percentage of change of spoor frequency and CV with increase on sampling effort.

Objective 4: In order for any index to be considered ideal, besides depicting a linear, positive, monotonic relationship with a slope equal to a true density estimate (Overton 1980; Lancia *et al.* 1994; Conroy 1996; Poole *et al.* 2003), the index must also be sensitive to actual changes. Changes detected by an index should be a reflection of changes on true abundance irrespective of the level of change (e.g. it should not only be able to detect changes above 50% or more). Unlike other studies designed specifically for calibrating an index (Vincent *et al.* 1991; Strayer 1999; Poole *et al.* 2003; Pollock 2006), which allowed for population manipulation (i.e. removal) as means of ascertaining the index sensitivity (index ability to detect true changes) at different population levels (e.g. N=100, 80 or 20), this study was not primarily designed for calibration purposes. Being opportunistic, more in-depth analysis of how spoor tracking would react under different population density levels as described by Strayer (1999) and Pollock (2006) could not be employed. Thus, Vincent *et al.* (1991) Kilometer Index Sensitivity Method of plotting the percentage change in the annual spoor and radio-tracking density estimates against time (years) was more appropriate for this study.

5.2 Interpretation and application

Literature provided a framework for the interpretation of results. Overton (1980) defined an ideal index as a constant-proportion type index, which was later defined as an index that depicts a linear, positive, monotonic relationship to a true abundance estimate (Conroy 1996), has a slope equal to one (Poole *et al.* 2003) and an intercept of zero (Lancia *et al.* 1994). Caughley (1977) stipulated that this type of relationship between index and true density estimate is seldom found due to the non-randomness of the distribution of biodiversity. Both Dice (1948) and Conroy (1996) supported this conclusion. The second objective of this study relates directly to these predictions and findings. Due to the limited amount of data available (cheetah spoor) for the study only a weak to moderate positive linear relationship is expected, with the regression coefficients deviating from the theoretical values. Nevertheless, the data set used for this study was identified as ideal for analysis during a cheetah census workshop in Tanzania (Funston, Houser, Forster & Smith 2004).

The outcome of the study will at least guide further research for developing a useful population index. In the case of a positive relationship between spoor and telemetry density estimates, recommendations on how to improve the sampling design, by incorporating cheetah ecology and behaviour, would be made. Preliminary results for determining the optimum sampling size to reduce spoor frequency variability would also allow recommendations to be made, as would the sensitivity analysis for detecting annual change in density.

The lack of a true density estimate (the absolute abundance), for the study area places a limitation on the ability to assess the true bias of each technique. However, true density estimates seldom exist for carnivores (Gese 2001) unless long-term studies are conducted (Funston *et al.* 2001). Therefore, the absence of this value should not constitute a hindrance to comparing how other relative abundance techniques perform, provided their assumptions are acknowledged and contained as far as possible. Furthermore, population trends are of greater importance for management objectives (Dice 1941, Pollock *et al.* 2002) for which precision rather than accuracy are vital. In addition, comparison should be made whenever one method can be shown to provide a more reliable estimate than the other. For example, radio telemetry was considered superior to spoor frequency in this study because the number of individuals occupying a certain area was known with certainty during the study period. Thus, it provides an ideal proxy to compare density estimated using spoor frequency. In summary, a comparison between these relative methods only provides an indication of how their estimates compare over time.

5.3 Conclusion

The selection of a study approach is influenced by the quality and relevance of reviewed literature on a particular technique in relation to the researcher's study objectives. The analytical framework for this study was primarily based on three key published papers consisting of Vincent *et al.* (1991), Stander (1998) and Funston *et* *al.* (2001). In addition, other pertinent literature on indices of abundance and radio telemetry were consulted. The annual minimum cheetah density estimated by Marker (2002) using radio telemetry provided an independent density estimate against which cheetah density from spoor tracking was compared. This approach to compare radio telemetry with an index is widely applied in the wildlife management literature (van Dyke *et al.* 1986; Vicent *et al.* 1991; Smallwood & Fitzhugh 1995; Stander 1998; Funston *et al.* 2001; Kunkel *et al.* 2005; Gese *Undated*). The certainty of knowing which individual is recorded within the sampling area gives the population estimate far more accuracy than many other methods of assessing density. It is for these reasons that radio telemetry was used as the independent variable during the analyses. Spoor tracking, being an index, requires evaluation to ascertain its potential as a monitoring technique to determine the relative or absolute cheetah abundance within the study area. In general there is lack of resources to conduct intensive research. Finding alternative techniques which could yield reliable results that are less expensive could be an ideal approach in wildlife management. Spoor tracking could be a useful technique for use by CCF.

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COMPONENT B

EVALUATION OF SPOOR TRACKING TO MONITOR CHEETAH ABUNDANCE IN CENTRAL NORTHERN NAMIBIA

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Evaluation of spoor tracking to monitor cheetah abundance in Central Northern Namibia

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The use of indirect indices, such as the detection of tracks (spoor), to estimate animal abundance has been questioned due to the impact of environmental variables such as substrate quality. It is important, therefore to calibrate indices against absolute abundance estimates made using other techniques. It is also important to determine whether indices are responsive to annual changes in population, and to understand the relationship between sampling effort and population estimates. In this study, spoor density for cheetah was evaluated against density estimated using radio telemetry on a commercial farm in Namibia. Least-linear regression and Spearman's correlation were used to evaluate the relationship between density estimates derived by the two methods. Two transects, one situated within medium to thick bush (A) and the other on mostly open vegetation (B), showed no significant relationship with density as estimated through radio-telemetry. Transect A showed a weak positive trend whereas B showed a negative trend. Spoor data pooled across the two transects also showed no significant relationship with estimated density (r^2=17.8, Y=2.05X+0.35). Only transect A showed a similar pattern to the radio telemetry in terms of annual density changes. This study established that a minimum sampling effort of 1655km is required in order for spoor tracking to minimize variability in spoor frequency. Differences in spoor tracking and radio-telemetry sampling areas sizes, the lack of more localized absolute abundance estimates, the spatial and social organization of cheetah and the fact that transects were originally

laid for ungulate strip counts, were factors suggested to have contributed to the poor calibration results obtained in this study. Further research is recommend, in which transects are reconfigured to encompass home ranges of as many individuals as possible, and in which attempts are made to identify individuals responsible for spoor possible through the use of remote-cameras placed along the transects.

Key words: cheetah, evaluation, calibration, abundance, spoor tracking, radio tracking

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INTRODUCTION

The foundation for the development and design of sound conservation practices and of efficient wildlife management is dependent largely on knowing a species' population status and changes in this over time. A variety of sampling techniques is available to either directly or indirectly ascertain species' density (reviews Lancia *et al.* 1994; Gese 2001; Wilson & Delahay 2001; Sadlier *et al.* 2004). Each technique has advantages and disadvantages (see Lettink & Armstrong 2003 for a review of census selection criteria).

Direct sampling strategies utilized for population estimates of carnivores include mark-recapture on lions (*Panthera leo*) (Castley *et al.* 2002), and radio telemetry on cheetahs (*Acinonyx jubatus*) (Gros *et al.* 1996; Marker 2002), leopards (*Panthera pardus*) (Stander 1998) and lions (Funston *et al.* 2001). Radio telemetry was utilized on cheetahs mostly for home range and territory determination (Caro 1994; Purchase & du Toit 2000; Marker 2002; Broomhall *et al.* 2003). Caro (1999) used the line transect method to estimate lion and spotted hyena (*Crocuta crocuta*) densities in woodland habitats while Bowland (1994) and Kelly *et al.* (1998) used systematic and random photographs taken by tourists and park personal to estimate cheetah density in the Kruger National Park and Serengeti, respectively.

For elusive, difficult to spot carnivores like cheetah and leopard, the financial costs required to acquire sufficient statistical samples renders commonly used methods unsuitable for population estimates, unless finances are not a limiting factor. Caro (1994) identified sample size as a limiting factor for using direct methods of census for cheetah in his study in the Seregenti. Moreover, an absolute abundance value, a precise value of the number of individuals of a species in a given area at a particular time (Caughley 1977), may not always be necessary (Dice 1941) for either scientific or management objectives (review Yoccoz *et al.* 2001; Pollock *et al.* 2002). The tendency for carnivores to make use of roads (Koford 1978; Karanth 1995; Broomhall *et al.* 2003) provides an opportunity for the use of indirect methods as a census technique.

Indirect methods rely on the presence and detectability of signs of an animal's presence (Wemmer *et al.* 1996). The presence of tracks along transects has been widely investigated for different species. Dzieciolowski (1976), van Dyke *et al.* (1986) and Smallwood & Fitzhugh (1995) used this method on mountain lion (*Felis concolor*), Funston *et al.* (2001) employed this methodology specifically for lions, whereas Stander (1998) used it on leopard. Gusset and Burgener (2005), instead used this method with spoor measurement to estimate density of leopard, caracal (*Caracal caracal*), serval (*Leptailurus serval*), African wildcat (*Felis silvestris libyca*), brown hyena and black-backed jackal (*Canis mesomelas*). These studies made reference to the appropriateness of this method for other carnivores like cheetah as well as for ungulates. Mahon *et al.* (1998) also evaluated the use of spoor tracking against random sandplots and spotlight counts for feral cats (*Felis catus*), red foxes (*Vulpes vulpes*) and dingoes (*Canis lupus dingo*). Other examples of indices include scent stations with sand plots by Harrison (1997) for jaguar (*Panthera onca*), jaguarandi (*Felis yagouarundi*), ocelot (*Felis pardalis*) and puma (*Puma concolor*), auditory indices by Ogutu & Dublin (1998) for lions and spotted hyenas, pellet counts or molecular scatology for red foxes (Cavallini 1994; Webbon *et al.* 2004), ground transect surveys of physical structures such as a count of black-tailed prairie dog (*Cynomys ludovivianus*) burrows (Mallick *et al.* 1997; Severson & Plumb 1998), and interviews and questionnaires for cheetah (Joubert & Mostert 1975; Myers 1975; Gros *et al.* 1996; Marker-Kraus *et al.* 1996; Gros 1998; Gros & Rejmanek 1999). The major assumption is that an ideal frequency index is a constant-proportion type index (Overton 1980).

Indices have been criticized due to factors that bias counts, and thus influence estimates (Sutherland 1996). Such factors include substrate suitability (Cavallini 1994; Mallick *et al.* 1997; Sadlier *et al.* 2004) in relation to detection probability (Mallick *et al.* 1997; Funston 2005 *pers. com.*), environmental heterogeneity, topography and vegetation, animal movement patterns (Stern 1998) and distribution (Dice 1948), factors influencing the decay rate of signs (weather, time of day, density of other animals) (Wemmer *et al.* 1996; Stern 1998), the effect of roads (Servin *et al.* 1987) or their positioning (within an animal home range (Burt 1943)) on animals' normal behaviour (Mahon *et al.* 1998; Wiesel 2005 *pers. com.*) as well as the road length (see Dzieciolowski 1976; Stander 1998; Funston *et al.* 2001) and orientation (Smallwood & Fitzhugh 1995). Error in observation is another cause for concern when using indices (Smallwood & Fitzhugh 1995).

Index calibration through double sampling (Eberhardt & Simmons 1987; Pollock *et al.* 2002) prior to the use of indirect census methods as part of a population monitoring program (Overton 1980; Ford 2000) is proposed as a means of accounting for some of the factors listed above. Calibration needs to be site specific and performed over time to account for seasonal differences (Walker *et al.* 2000) as well as vegetation structure. To validate the effectiveness of an index the following must be tested: (1) the assumption of constancy, and (2) the ability of an index to detect and establish the magnitude of changes over time (sensitivity analysis after Vincent *et al.* 1996). Depending on the results of (1) and (2), an index can be used to calculate an estimate of relative abundance (Eberhardt & Simmons 1987). Greenwood (1996) also proposed sampling protocol standardization as a means of addressing some of the problems with indirect population census techniques.

In Namibia leopard and wild dog (*Lycaon pictus*) spoor counts were calibrated by Stander (1998) in the Kuadom Nature Reserve, and this was also done for lions by Funston *et al.* (2001) in the Kgalagadi Transfrontier Park. Funston (2005 *per. com.*) also carried out calibration for population census methods for cheetahs and lions in the Serengeti, Tanzania. In these three studies, absolute abundance was estimated from a combination of methods, namely: the marking of individuals for identification during observations, radio telemetry, and call up stations. These calibrations, with the exception of the one in Serengeti, resulted in high correlation in methods and precision in population estimates. However, relatively few calibrations for census methods have been performed because of the difficulties of obtaining true density estimates for many carnivores (Gese 2001). No empirical study was found in the literature focusing on the calibration of cheetah spoor for estimating wild cheetah relative or absolute abundance in Namibia.

The Namibian cheetah is classified as a protected game species under the Nature Conservation Ordinance of 1975 (Nowell 1996) and as Vulnerable under Appendix I of CITES with an annual CITES quota of 150 individuals (Caro 1994; Nowell & Jackson 1996). Namibia harbours the largest free ranging cheetah population in the world with 95% of the population existing on commercial livestock and game farmlands (Marker 2002) in the north central area of the country. However, like many wide-ranging carnivore species, there are no scientific based population estimate (Sunquist & Sunquist 2002). Morsbach's (1986) country-wide cheetah abundance estimate of 2500 is frequently referenced, but a more recent estimate using sighting reports and questionnaires has estimated an abundance for the country of between 3138 and 5775 (Stander 2004). Sighting reports in Namibia have been suggested to be inflated due to autocorrelation (Nowell & Jackson 1996), a direct result of cheetah being a highly mobile species as well as difficulties of local communities in differentiating cheetah from leopard. Questionnaires present similar problems in population estimates and both techniques are more beneficial for acquiring base-line information on presence and absence of a species (Marker *et al.* 2004).

The Cheetah Conservation Fund (CCF), a Namibian non-profit organization, focuses its efforts on reducing human-cheetah conflict within farmlands, while enhancing the current understanding of the species through ecological and biological research (Marker 2002, see www.cheetah.org). In Namibia, cheetahs have been persecuted through killing or live capture and removal due to perceived or actual predation on wildlife and/or livestock (Nowell & Jackson 1996; Marker *et al.* 2003). Although there has been a lot of research conducted on Namibia's cheetah, including rates and reasons for removals and demographics of those handled and removed, and baseline radio-tracking data (Marker *et al.* 2003) the true impact of these removals on the wild cheetah population remains unknown.

Building on CCF's long-term cheetah research programme, the study presented here used CCF's minimum annual density estimated using telemetry to compare another census technique which could provide a repeatable, cost-effective and precise relative abundance estimate for cheetah. The minimum density is used as a representation of true density estimate. The aim of the study, therefore, was to investigate the relationship between the densities ascertained via spoor frequency and radio telemetry, in order to determine the effectiveness of spoor counts as a method for population census and monitoring in wild Namibian cheetah. The following four objectives were identified:

1. To determine cheetah density per unit of effort (kilometers) and area (square kilometers) based on spoor frequency, using the spoor tracking census method.
2. To investigate the relationship between cheetah densities measured from two indirect census techniques (spoor tracking and radio telemetry).
3. To determine the optimum sampling effort required to obtain reliable spoor frequency counts (determine sampling effort in kilometers required to minimise variability in counts).
4. To determine the sensitivity of the spoor frequency census method to actual annual changes in cheetah density as measured by radio telemetry.

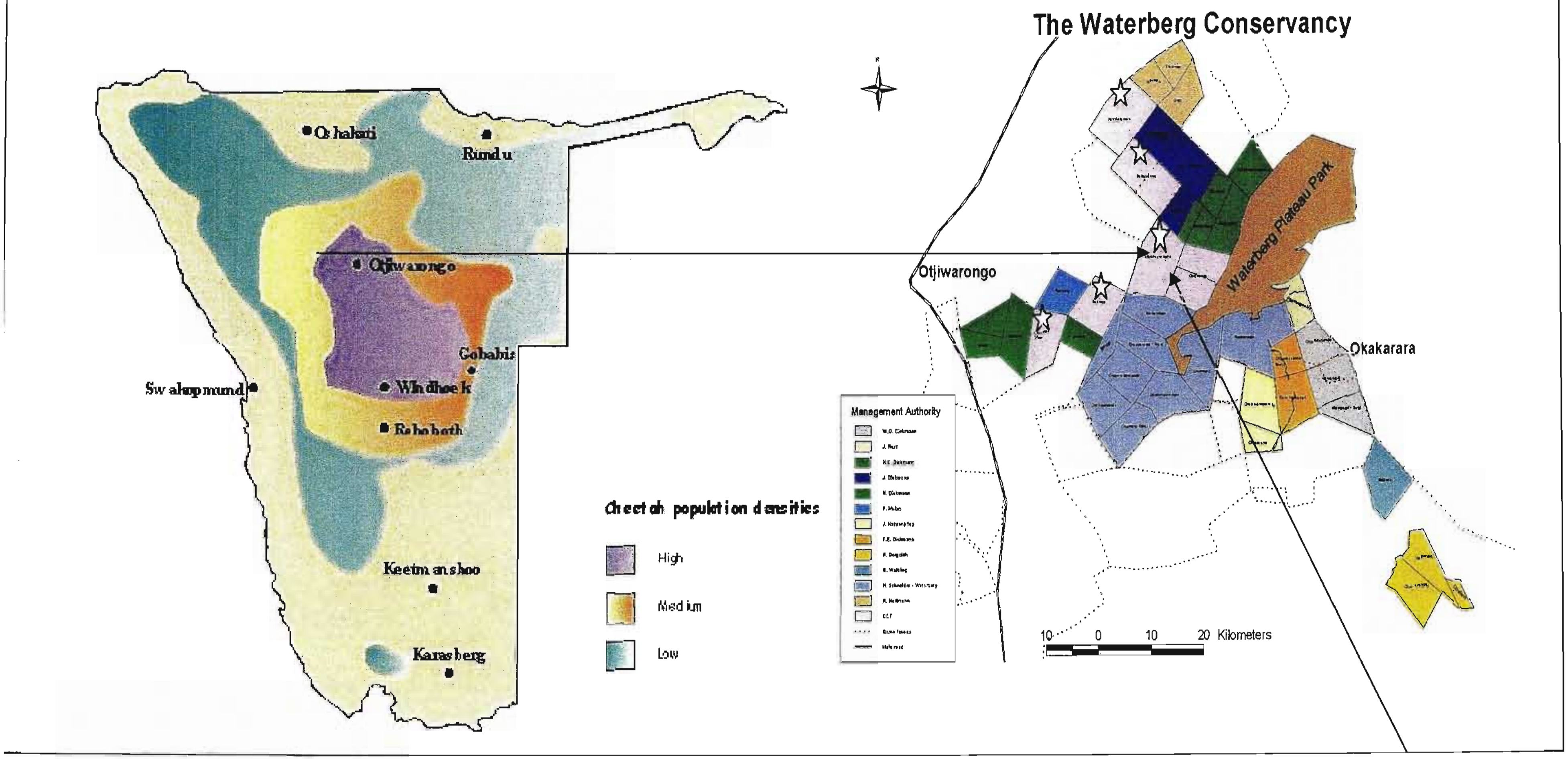
There were two major limitations to the study: (1) true abundance at a micro level, the level of the farm where spoor tracking was conducted, was unknown thus "comparisons between the different density estimates cannot identify which, if either, most closely reflects the reality" (Vincent *et al.* 1991), thus accuracy cannot be ascertained and (2) the lack of ascription of spoor to individuals over time limits the application of other density analytical techniques (e.g. mark-recapture). Consequently, a comparison between estimates derived from two relative techniques only indicates how these estimate relate to each. Nevertheless, true density estimates for carnivores are seldom available (Gese 2001) thus comparisons are encouraged particularly when one of the methods is considered to be more accurate than the other. The results presented here should, therefore be viewed as a foundation for future spoor tracking research on free-ranging cheetah populations in the study area.

STUDY AREA

The study was conducted in north-central Namibia, primarily within the Waterberg Conservancy (Fig. 1). A total area of 17 928km^2 (19°30'S to 23°30'S and 16°E to 19°E) was sampled during radio-telemetry whereas spoor tracking was conducted on a single farm, Elandsvreugde (16°39'0"E and 20°28'12"S), situated 44km east of Otjiwarongo with a total surface area of 73km^2 (Fig. 1). In general the area is geologically characterized by lithosols soils (Barnard 1998; FAO 1974) belonging to the Omingonde and Etjo formation (Schneider 1993), and topographically is generally flat with the exception of the Waterberg Plateau (i.e. 1800m above sea level). The area is semi-arid and lies between 400mm and 500mm rainfall isopleths (Barnard 1998). It is characterized by three major annual seasons: a wet-hot season (January – April), a dry-cold season (May – August), and an intermediate season (September – December), with an mean of 472mm ($\pm$ 156.28) of rain for the period (1993-2000).

The area is situated in the Thornbush Savannah vegetation zone, with dominant woody plant genera consisting of *Acacia, Dichrostachys, Grewia, Terminalia* and *Boscia* (Geiss 1971). Bush encroachment is a common phenomenon in the conservancy, manifested mostly by the dominance of *Acacia mellifera* (blackthorn), *Acacia tortilis* (red umbrella thorn) as well as *Dichrostachys cinerea* (sickle bush) (Barnard 1998) with little grass cover (Bester 1996). Open fields and transformed areas are prominent in the area as a result of previous land use forms (maize production) versus present usage (i.e. game and livestock farming) (Marker 2002). Elandsvreugde was subdivided into six vegetation categories depending on the percentage of canopy cover by Muroua *et al.* (submitted) (Table 1) as determined by aerial photography and ground observations.

The study area harbours a diversity of wildlife. Ungulates including oryx (*Oryx gazelle*), kudu (*Tragelaphus strepsiceros*), red hartebeest (*Alcelaphus buselaphus*), eland (*Taurotragus oryx*), duiker (*Cephalophus sp.*), springbok (*Antidorcas marsupialis*), steenbok (*Raphicerus campestris*), and warthog (*Phacochoerus africanus*) and carnivores such as brown hyena (*Hyaena brunnea*), caracal (*Lynx caracal*), jackal (*Canis mesomeles*) and leopard occur in the area.



Cheetah Conservation Fund,
Research and Education Centre:
Elandsvreugde Farm

Fig. 1. The Namibia cheetah distribution and position of the Waterberg Conservancy and the Cheetah Conservation Fund (CCF) research center and neighboring farms (☆). Adapted from Marker (2002) and Hasheela (2004).

MATERIALS AND METHODS

Radio tracking density analysis

The minimum cheetah density for the study area, determined via telemetry by Marker (2002), was used in this study to evaluate spoor tracking as a possible surveying technique to monitor cheetah abundance. This was determined by dividing the annual radio-tracking (RT) area by the number of collared cheetahs in groups known to be in the area that year (Marker 2002). Group density of collared cheetahs was considered by Marker (2002) to be more representative of the true density in the study area than for single tracked animals, thus its use for comparisons. A full description of the RT sampling and analysis protocols is provided by Marker (2002). Only the 1995 to 2000 annual RT densities are used in this study.

Spoor tracking sampling and statistical analysis

Spoor frequency was investigated as a suitable index of the true population density and is defined as the number of kilometers per spoor on the transect, whereas the number of individual spoor was used as an indicator of density (Stander 1998; Funston *et al.* 2001). Sampling effort referred to distance in kilometers surveyed, and the coefficient of variance (CV) was used as the measure of spoor frequency variability (Stander 1998; Gibbs 2000; Funston *et al.* 2001;) to determine its applicability to monitor long-term population trends (Boyes & Perrin 2006). Index sensitivity was defined as the ability of the index to monitor population trends over time (Vincent *et al.* 1996).

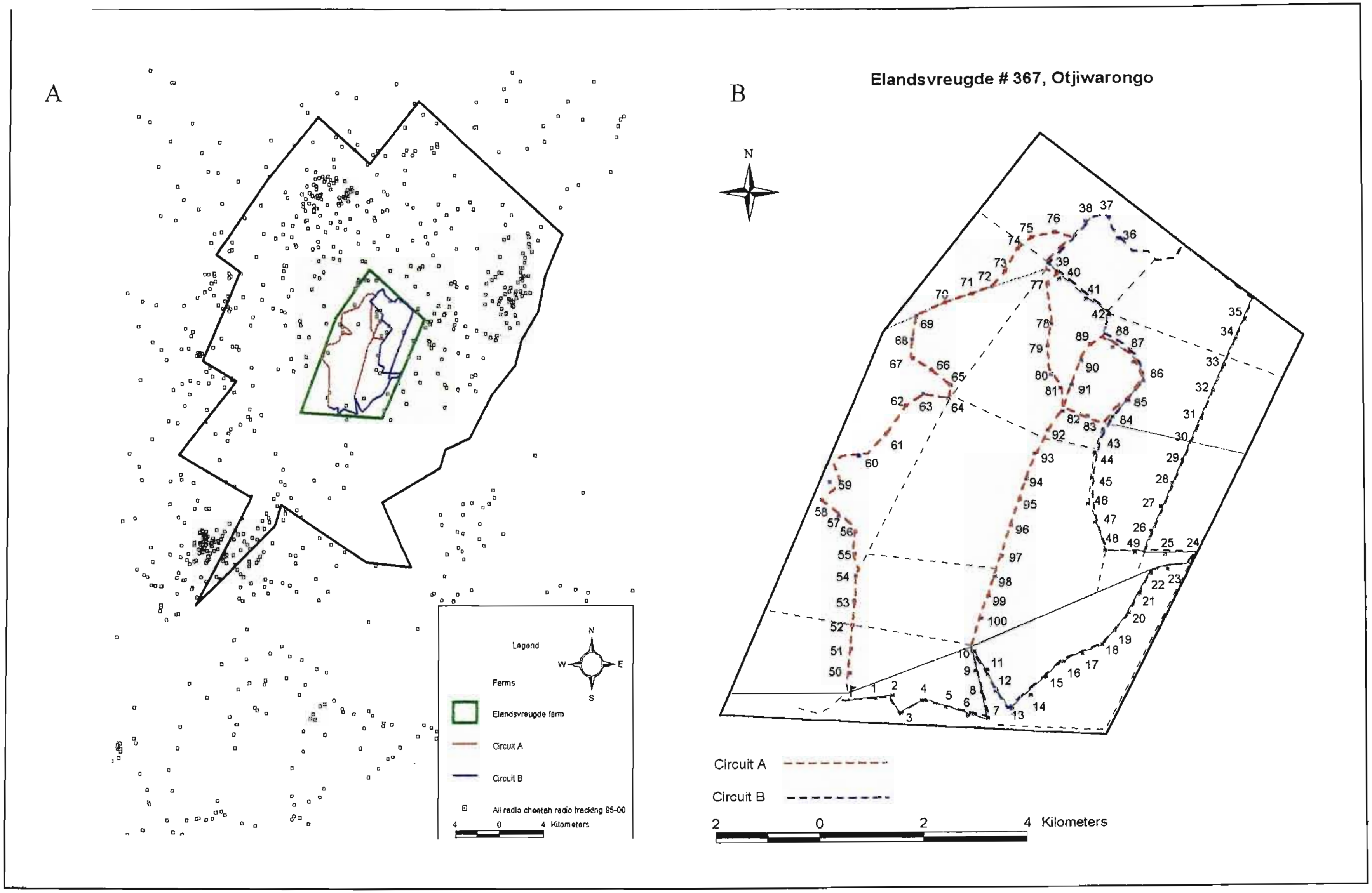


Fig. 2.a The radio tracking sampling area, with corresponding fixes, for the 1995-2000 period. Highlighted are the spatial boundaries for the radio and spoor tracking methods, and the spoor tracking transects. **2**.b Enlarged map of the spoor tracking zone. Numbers represent established sample points, 0 to 49 and 50 to 100 in the transects A and B, respectively.

Spoor tracking: data collection procedure

Spoor tracking was conducted on two designated farm routes (Fig. 2b) which traverse through the different vegetation types on the farm. The farm routes were defined as transect A and B which were 26.64 km and 28.54 km in length respectively. Spoor tracking was conducted monthly over a five-year period (1995 to 2000) between 8H30 – 11H00. Thus, 55.18 km were surveyed monthly and equaled a total of 3482.64 km covered over 63 months. Transects were demarcated with numbered metal poles located 0.53±.02 km apart on transect A, and 0.60±.08 km on transect B. The two transects are adjacent to each other with some overall and a maximum distance of approximately 6 km separating the furthest points (Fig. 2). Serial correlation as well as autocorrelation was anticipated due to this proximity. Transect A was predominantly within bush vegetation (common bush I, and II and thick bush) whereas transect B was mostly open vegetation (savannah, open field and transition zones) (Table 1).

Table 1. Vegetation zones in the Elandsvreugde farm (7300ha, spoor tracking study area) with corresponding canopy cover and percentage of the total farm covered by a vegetation type plus the linear combined (transect A and B) distances (km) surrounded by a particular vegetation type and the respective percentage of the total transect length (adapted from Muroua *et al.*, submitted).

Vegetation types	Canopy cover	Area covered (% of entire area)	Linear distance (km)	% total transect length
Common bush I	16 – 30%	1299.4ha (17.8%)	13	25.24
Common bush II	31 – 75%	2737.5ha (37.5%)	14	27.18
Open field	0-5%	1627.9ha (22.3%)	11	21.36
Transition area	40 – 80%	459.9ha (6.3%)	3	5.83
Thick bush	76 – 100%	781.1ha (10.7%)	4	8.74
Savannah	6-15%	394.2ha (5.4%)	6	11.65

A vehicle (initially a Toyota Venture model 1995 and alter an Isuzu model 1998) was driven at constant speed of 10km/hour, with a tracker seated on the bonnet. Upon spoor detection the vehicle was halted, and the spoor was verified. In addition to this, the associated odometer reading or sample area, the direction in which the animal was heading, and a count of the spoor, were recorded. Only the most recent spoor, i.e. assumed to be spoor from the night prior to tracking due to its clarity, and with a high degree of certainty of being cheetah spoor were recorded, as the intent was to derive an estimate of the current density (Rabinowitz 1997).

Spoor tracking statistical analyses

Cheetah spoor frequency was determined by dividing the number of kilometers by the cumulative number of samples with fresh spoor recorded; while cheetah density was calculated by dividing the total number of spoor by total transect length. Spoor densities are expressed per 100km. These were tested for normality and homogeneity using the Kolmogorov-Sminorv and Levene's test respectively (Durrheim 2002).

The least-linear regression was used to determine the nature of the relationships between sample effort (distances in km), or density determined via radio telemetry, as independent variables, with either spoor frequency or spoor density as the dependent variables. The non-parametric Spearman's correlation estimate was used to indicate level of strength of relationships. A significant positive correlation refers to the evidence that the quantifiable strength of the relationship between variables is positive and not due to random chance only that allows for prediction of independent variable from the dependent variable. The above calculations and analyses, i.e., spoor frequency, spoor density, least linear regression, and Spearman's correlation coefficient were performed for each individual transect, and for an overall spoor frequency and density (i.e. data pooled for the two transects). The Kruskal-Wallis test (KW) was used to compare annual frequencies and densities within transects while the Mann-Whitney test (U) was used to compare annual frequencies and densities between transects. All tests were two-tailed, significance was determined at 95% ($P < 0.05$), and averages are reported with associated standard errors as a measure of precision.

The overall optimum sampling effort required to minimize spoor frequency variability was determined using bootstrap analysis (SIMSTAT 2.5 Trial Version 2004). This effort level was determined using the combined cumulative spoor frequency which represented the initial sample, from which samples of different sizes were randomly selected. Random sample selection was initiated by a random seed number, altered per interaction, where sample size was increased progressively to 110. A sample of a particular size was replicated 500 times, and had the mean, 95% confidence interval and standard error reported. After determining the CV (standard deviation divided by the sample means of the counts (Gibbs 2000; Funston *et al.* 2001; Saltz *et al.* 2004)), both this and the spoor frequency were regressed against sample effort. At the point where no further gain in precision was achieved this level was regarded as the optimum sampling intensity (after Stander 1998 & Funston *et al.* 2001). This determination is subjective (Greig-Smith 1983) and guided by the decrease in the percentage of change of spoor frequency and CV with increase on sampling effort. Index sensitivity was determined by plotting the annual percentage change in density as estimated by each technique against time (Vincent *et al.* 1996). SPSS version 11.5 (2003) and Excel (2003) were used for statistical analyses.

RESULTS

Radio tracking density estimates

The mean minimum cheetah density for radio tracked animals in groups for the 1995 to 2000 period was 2.50 ($\pm$0.78) per 1000 km^2, with a mean of 16 ($\pm$4.73) cheetahs in groups tracked per year (Table 2).

Table 2. Annual number of cheetahs in groups radio-collared from 1995 to 2000, with corresponding annual tracked surface area (km^2) and minimum group densities.

Year	No. cheetahs radio tracked in groups	Radio tracking area km^2	Minimum density of radio-collared cheetahs in groups per 1000 km^2
1995	21	7416.1	2.83
1996	18	5263.6	3.42
1997	17	5663.2	3
1998	17	7390	2.3
1999	16	7138.1	2.24
2000	7	5911.2	1.18
Total	**96.00**	**38782.20**	**14.97**
Mean	**16.00**	**6463.70**	**2.50**
Standard error	**4.73**	**959.81**	**0.78**

Spoor tracking: frequencies and densities

Transect A

A total of 63 samples, equivalent to 1678.32 km, with an annual mean of 279.72 ± 74.88 km were surveyed along this transect over the five year period (Table 3). In total, 21 fresh spoor were encountered from 16 samples. Therefore, the transect had an overall spoor density of 1.25/100 km with a mean of 1.13(±0.94) per annum and an overall frequency of 104.90 km/spoor with an mean of 100.85(±65.20) per annum (Table 3). The frequencies and densities showed significant variation over time, KW X^2=12.8, *P* =.025 and KW X^2=11.4, *P* =.044 (df. 5), respectively.

Table 3. Cheetah spoor along transect A from 1995 to 2000, showing annual sample sizes, sampling effort (km), frequencies (km/spoor) and densities (spoor/km) per 100 km with totals, means, standard errors (SE) and 95% confidence intervals (95%CI).

Year	N (transects)	Sampling effort (km)	No. of transects (samples) with fresh spoor	Frequencies (km/spoor)	Total spoor count	Densities/ 100 km
1995	5	133.2	0	0	0	0
1996	12	319.68	7	45.67 (15.54±13.72)	9	2.82 (2.82±2.83)
1997	10	266.4	2	133.2 (5.33±11.23)	3	1.13 (1.13±2.53)
1998	12	319.68	2	159.84 (4.44±10.37)	4	1.25 (1.25±2.92)
1999	12	319.68	3	106.56 (6.66±12.05)	3	0.94 (0.94±1.70)
2000	12	319.68	2	159.84 (4.44±10.37)	2	0.63 (0.63±1.46)
Total	**63.00**	**1678.32**	**16.00**		**21.00**	
Mean	**10.50**	**279.72**	**2.67**	**104.90**	**3.50**	**1.25**
SE	**2.81**	**74.88**	**2.34**		**3.02**	
95% CI	**2.25**	**59.91**	**1.87**		**2.41**	

Transect B

Transect B had a higher sampling effort of 1798.02km than transect A, with an annual sampling effort mean and standard error of 299.67±80.22 km. However, the mean of this distance was not significantly different to transect A (t =-.468, *P* =.650, df. 10). Despite the higher sampling effort, only on eleven occasions or 17% of the total number of samples (45% lower than for transect A), was cheetah spoor encountered, resulting in a high overall transect spoor frequency of 224.75 km/spoor with an annual mean of 180.81(±135.79) and a lower overall transect spoor density of 0.61/100 km with an mean of 0.62(±0.27) per annum (Table 4). In contrast to transect A, transect B's inter-annual frequencies and densities did not vary significantly (KW X^2=2.522, P =.773 and KW X^2=2.215, P =.819, df. 5) between years.

Table 4. Cheetah spoor along transect B from 1995 to 2000, showing annual sample sizes, sampling effort (km), frequencies and densities per 100 km with totals, means standard errors (SE) and 95% confidence intervals (95%CI).

Year	N (transects)	Sampling effort (km)	No. of transects (samples) with fresh spoor	Frequency (km/spoor)	Total spoor count	Densities/ 100 km
1995	5	142.7	1	142.70 (5.73±12.81)	1	0.70 (0.70±1.56)
1996	12	342.48	1	342.48 (2.39±8.27)	1	0.29 (0.29±1.01)
1997	10	285.4	1	285.40 (2.86±9.06)	2	0.70 (0.70±2.21)
1998	12	342.48	1	342.48 (2.39±8.27)	1	0.29 (0.29±1.01)
1999	12	342.48	1	342.48 (2.39±8.27)	3	0.88 (0.87±3.02)
2000	12	342.48	3	114.16 (7.16±12.95)	3	0.88 (0.87±1.58)
Total	**63**	**1798.02**	**8**		**11**	
Mean	**10.50**	**299.67**	**1.33**	**224.75**	**1.83**	**0.61**
SE	**2.81**	**80.22**	**0.82**		**0.98**	
95% CI	**0.69**	**19.81**	**0.20**		**0.24**	

Combined transect data

Overall, a total of 3482.64 km, (mean distance of 580.44±155.38 km per annum), was traversed, which provided an overall spoor frequency of 151.42 km/spoor (181.73±64.47) and overall spoor density of 0.92/100 km (0.86±0.37) per annum for the five-year period (Table 5). Neither the overall frequency (KW X^2=3.973, P =.55) nor density (KW X^2=3.515, P =.62) for the combined data indicated significant variation over time. Likewise, when transect frequencies (U=10, P =.2) and densities (U=10, P =.2) were compared no significant differences were evident.

Using a penetration rate of 1 km for every 1.32 km^2 (ratio of the spoor tracking farm size with the total spoor tracking length transect), overall converted annual spoor density for the five-year period was of 0.7/100 km^2 (0.15$\pm$0.06). The annual cheetah density per 100 km^2 based on spoor densities are presented in Table 6. Converted densities for the individuals transects were not significantly different from those estimated using radio telemetry (P =0.06 and 0.15). However, the overall density was (P =0.02).

Table 5. Overall cheetah spoor for combined transects (A and B) from 1995 to 2000, showing annual sample sizes, sampling effort (km), frequencies (km/spoor) and densities (spoor per 100 km with totals, means, standard errors (SE) and 95% confidence intervals (95%CI).

Year	N (transects)	Sampling effort (km)	No. of transects (samples) with fresh spoor	Frequencies (km/spoor)	Total spoor count	Densities/ 100 km
1995	5	276.4	1	276.40 (11.06$\pm$24.72)	1	0.36 (0.36$\pm$0.81)
1996	12	663.36	8	82.92 (32.25$\pm$28.47)	10	1.51 (1.51$\pm$1.70)
1997	10	552.8	3	184.27 (16.58$\pm$26.70)	5	0.9 (0.90$\pm$1.54)
1998	12	663.36	3	221.12 (13.82$\pm$25.00)	5	0.75 (0.75$\pm$1.43)
1999	12	663.36	4	165.84 (18.43$\pm$27.12)	6	0.9 (0.90$\pm$1.64)
2000	12	663.36	4	165.84 (18.43$\pm$27.12)	5	0.75 (0.75$\pm$1.21)
Total	**63**	**3482.64**	**23**		**32**	
Mean	**10.5**	**580.44**	**5.33**	**151.42**	**3.83**	**0.92**
SE	**2.811**	**155.38**	**2.88**		**2.32**	
95% CI	**0.694**	**124.32**	**2.30**		**1.85**	

Table 6. Cheetah estimated density from 1995 to 2000 per surface area (100 km^2) using a penetration rate of 1km: 1.32 km^2, based on transect A, B and overall spoor density estimates and the radio telemetry density estimates. Densities are presented with a mean and standard deviations.

	Transect A			Transect B			Transect A and B pooled			Telemetry density
Year	Distance (km)	Number of spoor	Density/ 100 km^2 *	Distance (km)	Number of spoor	Density/ 100 km^2 *	Distance (km)	Number of spoor	Density/ 100 km^2 *	(100 km^2)
1995	133.2	0	0.00	142.7	1	0.53 (0.53±1.19)	276.4	1	0.27 (0.27±0.61)	0.28
1996	319.68	9	2.13 (2.13±2.14)	342.48	1	0.22	663.36	10	0.11 (1.10±1.15)	0.34
1997	266.4	3	0.85 (0.85±1.92)	285.4	2	0.53	552.8	5	0.14 (0.55±1.23)	0.3
1998	319.68	4	0.95 (0.95±2.21)	342.48	1	0.22 (0.53±1.19)	663.36	5	0.11 (0.27±0.61)	0.23
1999	319.68	3	0.71 (0.71±1.29)	342.48	3	0.66 (0.53±1.19)	663.36	6	0.11 (1.10±1.15)	0.22
2000	319.68	2	0.47 (0.47±1.11)	342.48	3	0.66 (1.06±1.45)	663.36	5	0.11 (0.55±0.75)	0.12

* Density/100 km^2 = (((Number of spoor/Distance)*1)*100 km^2)/1.32 km^2

Density estimate comparisons: linear regressions and calibration

The individual transects and combined data density estimates were regressed against the density estimates determined via telemetry (Table 7, Fig. 3). Transect A showed a moderate correlation with telemetry density (Fig. 3A), but only 26% (r^2, Table 7) of the spoor density variation is explained by telemetry density, and the resulting model Y=6.28X-0.43. The density relationship is not significant (F=1.44, *P* =0.3). Both the slope and intercept 95% CI encompasses zero, providing relative evidence of these not being different to zero. Transect B, on the other hand, has a negative moderate relationship (Fig. 3C), with radio telemetry, the independent variable explaining 30% of spoor density variation (r^2, Table 7). As is the case for transect A, the slope is not significantly different from zero (t=-1.32, *P* =.26) with intercept 95% CI encompassing zero. As such, changes in spoor frequency along transect B do not reflect adequately changes in population determined by radio tracking. When data were pooled for the two transects and regressed (Fig. 3E), there was a positive (b=2.05) but poor linear relationship between the two density estimates (r^2=18%, Table 6). As for transects A and B, the relationship failed to reach significance (F=0.87, *P* =.41), with the regression coefficient not being different from zero, and with zero being encompassed by the intercept 95% CI (Table 7).

Table 7. Relationship between density estimates from radio tracking and spoor counts. Regression coefficients (N= sample number; a=intercept; b= rate of change; r^2= measure of variability prediction; SE = standard error; 95% CI = 95% confidence interval) and Mann-Whitnney test (F=test value; *P* = significance level) for annual transect A, B and combined data densities against radio telemetry density estimates.

Variables	N	a (95%CI)	b (95% CI)	r^2	SE (*100)	F	*P*
Telemetry vs. transect A	5	-0.43 (-4.19, 3.32)	6.28 (-8.27, 20.83)	26.4	90.3	1.43	0.3
Telemetry vs. transect B	5	1.09 (.06, 2.13)	-1.91 (-5.92, 2.1)	30.5	24.86	1.75	0.26
Telemetry vs. combined data	5	0.35 (-1.23, 1.93)	2.05 (-4.06, 8.16)	17.8	37.92	0.87	0.41

Similarly, the relationship between transect A and the overall spoor density expressed per area (km^2) (Table 6) were positive (b=4.76 and 0.22 respectively) but not significant ($F=1.45$, $P=0.3$ and $F=0.31$, $P=0.61$ respectively) (Fig. 3B, F). Transect B also had a non-significant relationship ($F=1.77$, $P=0.26$) but negative ($b=-1.45$) (Fig. 3D). Furthermore, their slopes are not significantly different to zero (transect A: $t=1.20$, $P=0.3$; transect B: $t=-1.32$, $P=0.26$; overall: $t=0.56$, $P=0.61$) with intercept 95% CI encompassing zero (transect A: -3.17, 2.51; transect B: 0.05, 1.61; overall: -0.20, 0.37).

Nevertheless, only the overall spoor density per area was significantly different to the radio telemetry estimates ($U=4$, $P=0.02$).

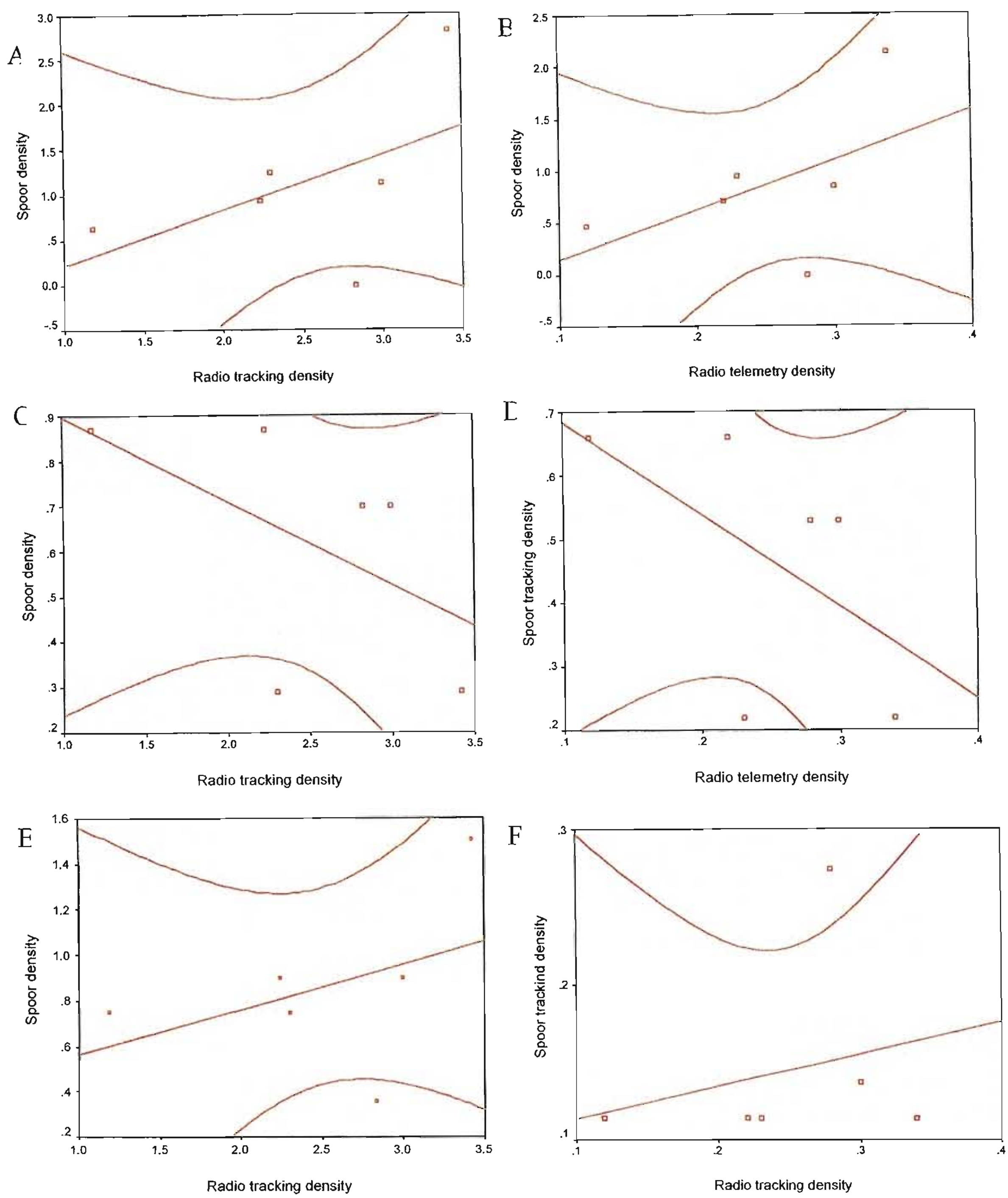


Fig. 3. Relationship between radio tracking density (per 100 km^2) and spoor density (per 100 km and 100 km^2) for transects A (A, B), B (C, D) and combined transects (E, F) determined using linear regression. The curved lines represent the 95% confidence interval (CI) for the best fit line. Drawing a horizontal line on (B,D and F) from the spoor density estimate through the regression and CI curve, down to the radio tracking density provides a calibration estimate of density (after Severson & Plumb 1998).

Determination of minimum sampling effort for spoor frequency estimates

Figures 4 and 5 present the spoor frequency and the coefficient of variation with sampling effort (km covered). The combined spoor frequency for transect A and B showed a drastic decrease with sampling effort increment, with further sampling resulting in low frequency at 30 samples (approximately 1655.4 km). Three major trends can be observed regarding the frequency at which spoor was encountered (Fig. 4, Table 8). The first phase, between 2 and 16 samples (827.7 km), is characterized by a drastic decrease in the distance required to encounter spoor or spoor frequency. The next two phases, phase II and III, starting from 17 to about 48 samples and from 49 to 63 samples, respectively, had few spoor recorded relative to phase I, despite these recording higher effort rates (1765.76 km and 717.34 km, respectively). The same pattern was followed for the analysis investigating the increase in precision with increased sampling effort (Fig. 5), with little gain in precision after thirty samples. The CV decreased only by 25.68% with the additional thirty samples (Fig. 5) despite a slight increase in spoor frequency of 25% at sixty samples. Thus, optimum sampling effort for this study was attained at a certain point during phase II or at approximately 1655 km.

Table 8. Three phases of spoor frequency counts with corresponding sample numbers (N), distances (km) and spoor frequency, with the mean and standard error (SE) for spoor frequency, coefficient of variance (CV) and its 95% confidence intervals (95% CI).

Phase	N	Distances (km)	Spoor frequency (km/spoor)	Spoor frequency (km/spoor) Mean± SE	CV	95% CI
I	15	827.7	75.25	128.84±104.17	80.85	114.86, 148.78
II	32	1765.76	126.13	102.62±34.16	33.29	120.20, 141.31
III	14	717.34	102.48	75.01±23.17	30.89	120.37, 139.88

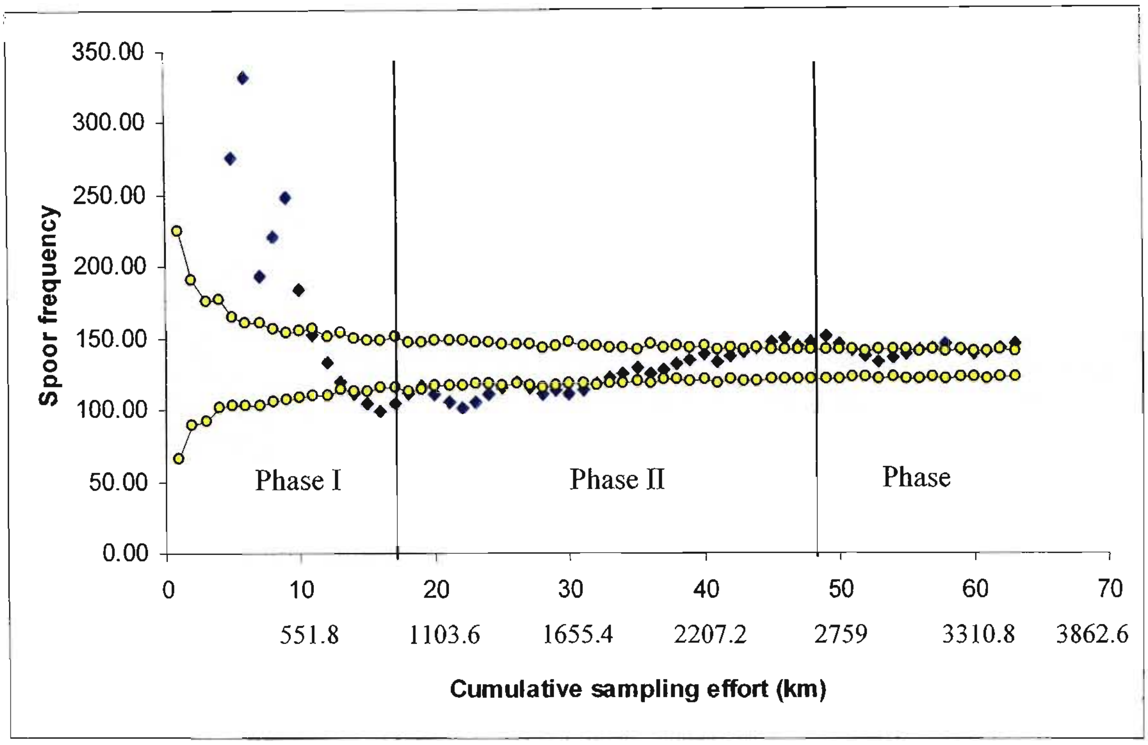


Fig. 4. Determination of minimum sampling effort required to estimate cheetah density using spoor tracking. Spoor frequency (number of km covered / number of spoor recorded) plotted against cumulative sampling effort (km). The diamond shaped dots represent the actual spoor frequency whereas the circles represent the corrected 95% confidence intervals determined using bootstrap analysis. Phase I, II and III depict the three stages of how spoor frequency variability changed with increase in sampling effort (n=63 samples equivalent to 3482.64 km).

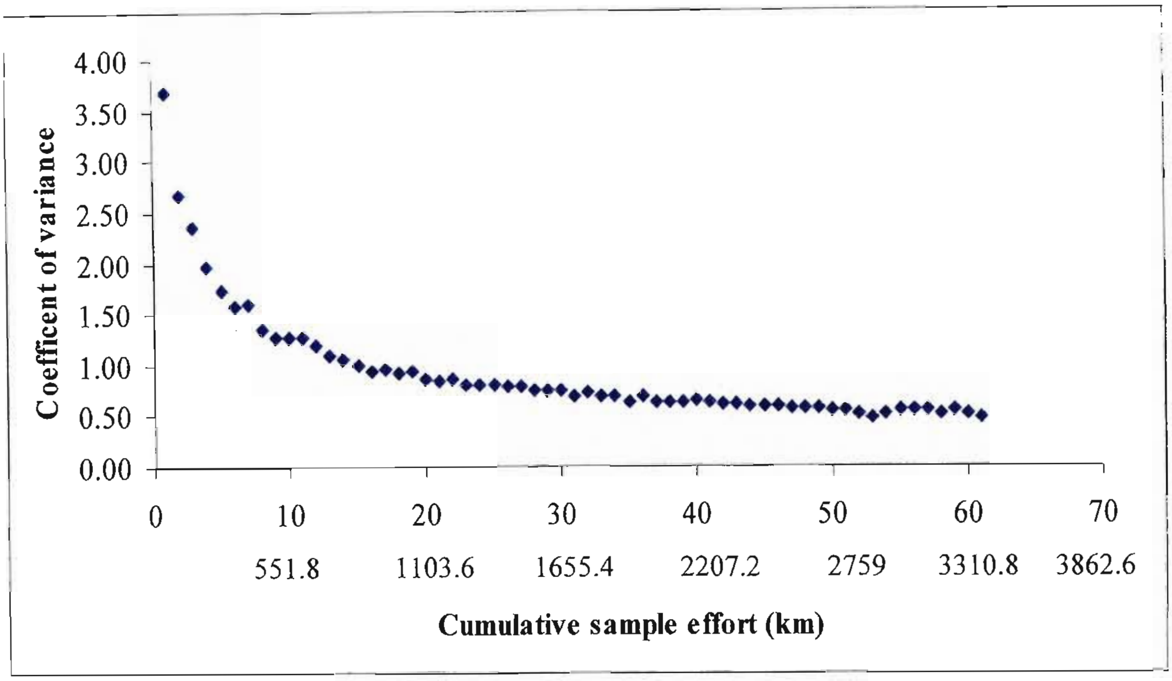


Fig. 5. Determination of minimum sampling effort required (distance in km) to reduce variability in cheetah spoor frequency estimated using spoor tracking. Coefficient of variation in spoor frequency plotted against cumulative sample effort (km). An asymptote was determined arbitrarily (Greig-Smith 1983) at about 1655 km (equivalent to 30 sample occasions).

Sensitivity of spoor tracking density estimates to annual changes in density as measured by radio tracking

Radio telemetry densities remained relatively consistent for the five years although they were lower in 2000 due to the lower number of animals that were tracked (Fig. 6, Table 2). When densities were expressed per 100 km, both transect A and the combined depict a similar pattern over time but at higher densities (Fig. 6 & 7). When densities were expressed per 100 km^2, only the combined transect showed a similar pattern but at lower densities than the telemetry. Transect B fluctuated on a yearly basis in both categories (per 100 km and km^2), but had similar densities in 1996 and 1998 (Fig. 6). The ability of spoor frequency to monitor relative cheetah abundance is presented in Fig. 7. When percentage changes were regressed against percentage change in the telemetry density, only transect A and the combined transects densities (per 100 km^2) showed a high correlation (r^2=0.71, P =.18 and r^2=0.77, P =0.13 respectively). Transect A (per 100 km) showed almost no relation (r^2=0.26, P =0.67) whereas the combined transect (100 km^2)

regression yielded an inverse relation (r^2=-0.60, P =0.28). On the other, density changes on transect B (expressed either per 100 km or 100 km^2) showed no relation to the telemetry changes (r_s=0.05, P =0.93 respectively).

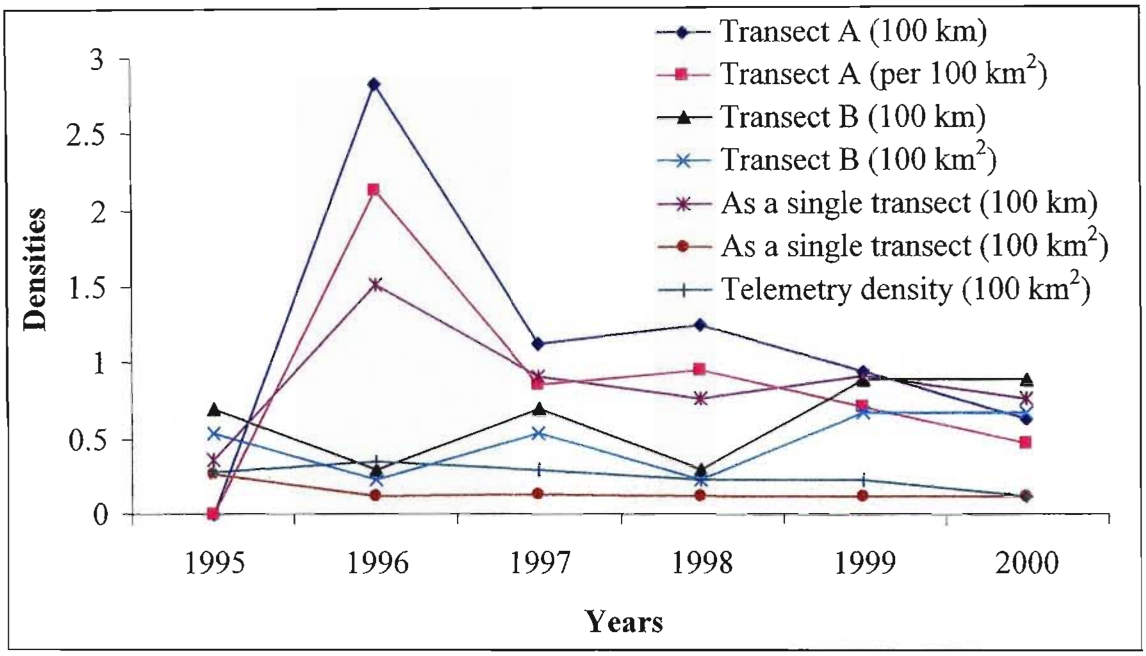


Fig. 6. Variability of radio telemetry (per 100 km^2) and spoor tracking densities per transect and combined (per 100 km and 100 km^2) by years.

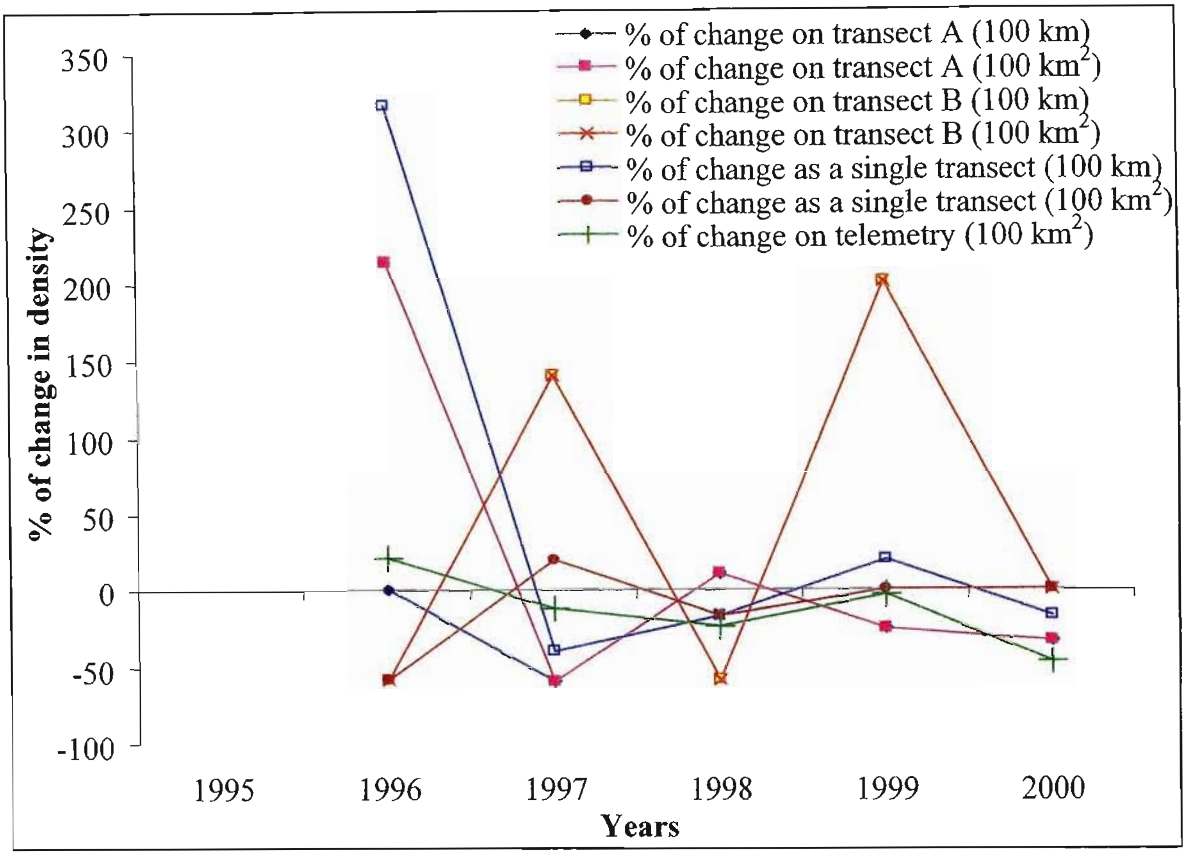


Fig. 7 Percentage change in spoor density (per 100 km and 100 km^2) from previous year (per transect and as a single transect) and telemetry densities (100 km^2) as a measure of index sensitivity. Negative values indicate a reduction in density estimates.

DISCUSSION

Linear regression and calibration

The primary objective of this study was to determine whether a relationship exists between spoor densities and densities obtained using radio telemetry. A non significant relationship was evident between the cheetah population density estimates provided by the two methods, despite the fact that a positive relationship was found between the data obtained from transect A and for data combined from both transects (i.e. single transect). Thus, a preliminary conclusion is that spoor frequency may not be regarded as a reliable method in terms of monitoring free ranging cheetah relative abundance. However, both the use of this method, evaluations and calibration of figures should be interpreted cautiously for various reasons.

The first reason for the lack of a strong correlation between the radio and spoor tracking methodologies is that they encompassed different spatial extents, which may have affected the results. The transects used for spoor tracking were originally laid out to conduct prey abundance counts and were circular rather than straight. Having straight transects would have ensured that different animals' home ranges were traversed during spoor tracking sampling. Thus, it is possible that spoor frequency density estimates reflect a home range, or part of it, and not the population in the radio tracking study area, despite some overlap in home ranges of individuals (Marker 2002) and the presence of transients. Skalski *et al.* (2005) postulates that weak inferences can be made or are present if sampling areas are a subset of the larger target population. Therefore, spoor frequency could indeed have application in monitoring cheetah relative abundance in the spoor tracking study area, but spoor tracking transect design needs to be modified.

A second reason for the weak correlation is a direct consequence of not having accounted for cheetah spatial organization and daily range during the planning and design of the transects. Wright (1991) advocated that strong correlations between methods are a result of chance because of the placement of transects in relation to the uneven distribution of animals. Similarly, Dice (1941) postulated that the placement of a transect in an aggregator area enhances correlations between an absolute abundance estimate and an index. Pollock (2006) also established that the power of presence-absence survey method to detect a decline in the population could be strengthened with the clumping of individuals at site. The farm Elandsvreugde (Fig. 2.a, b), where the spoor transects are located, appears to represent more of an excursion area, which is visited by cheetahs infrequently if compared to a core area site, thus the weak correlation.

A factor that may have influenced the results obtained for this study, is the substrate quality in relation to spoor detectability. Lithosoils are predominant in the spoor tracking sampling area (Barnard 1998) and these are fairly evenly distributed. Such soils are poorly formed resulting in a hard substrate (Buckman & Brady 1969) which presents a drawback in terms of spoor detectability, as highlighted by the tracker (Nghikembua 2005 *pers. com.*). As a result, some spoor may have been missed during sampling occasions. Funston (2005 *pers. com.*) suggested that the relationship between substrate quality and spoor detectability is vital.

The fourth possible explanation for the weak correlation between spoor and radio tracking results is related to the power of spoor as an index. The power of an index is dependent on the rate at which distinguishable signs are found, i.e. frequency. This means that transects used for spoor tracking must overlap with areas used by cheetah, and transects should therefore be set up with knowledge of cheetah movement patterns in the study area. Literature indicates that cheetah are known to follow roads (Sunquist & Sunquist 2002; Broomhall *et al.* 2003; Marker 2005 *pers. com.*), males scent-mark physical features located closely to open areas (Broomhall *et al.* 2003; personal observation) and females prefer more mosaic-type habitat (Marker 2002). The spoor transects were located within suitable cheetah habitat in terms of vegetation. The distribution of a species across a suitability habitat is determined by other factors besides vegetation such as prey abundance and other competitors (e.g. lions and spotted hyenas). Therefore, a low index power would be expected as other cheetah habitat suitability covariates were excluded during the transect design.

Furthermore, cheetahs within the study area have shown a certain degree of preference for medium to thick bush (Marker 2002) which coincides with kudu which according to Muroua *et al.* (*submitted*), is one of the preferred cheetah prey in the study area. Transect A, with surrounding vegetation composed mostly of medium to thick bush, had a higher spoor frequency than transect B. The existence of adjoining road networks could also have had an impact on the poor performance of the index. Monitoring of different roads could shed light on the use of roads and the areas used by cheetah. However, as advocated by Caughley and Sinclair (1996) and supported by Mahon *et al.* (1998), given that roads are rarely random samples of the habitat, changes in road-based indices may not necessarily reflect changes in density throughout the habitat.

The last explanation presented for the results obtained in this study is also related to spoor power. Rate-based indices are dependent on the rate at which signs are encountered (Jackson *et al.* 2005). Therefore, indices of abundance may be zero for species that live at low densities since there is a low likelihood of encountering signs (i.e. spoor). This phenomenon is referred to by Nichols and Conroy (1996) as "index bottom-out". The fact that in this study the 95% confidence intervals encompasses zero suggests that the spoor frequency may have been too low for cheetah to provide useful data.

Strayer (1999) reached a similar conclusion when investigating the power of the presence-absence survey method to detect population declines. The lower power was more prevalent for species that were rarely detected, or highly variable in their spatial distribution and for low sampling effort, these all translated to low encounter rates, and subsequently low power. The fact that cheetahs do occur at low population densities, are far-ranging and can travel in a single direction for extended distances corroborates the bottom-out concept. An alternative explanation for the low spoor power is that cheetahs in the study area do not make as much use of the roads as expected or that they use other farm roads adjoining the transects, which were not monitored. Also related to the bottom out concept is the possibility of spoor as an index having a threshold value beyond which changes in abundance are not detected (Powell 2000).

Evaluation and calibration results from this study are inconsistent with Stander (1998) and Funston *et al.* (2001) who found significant correlations between spoor frequency and true population densities estimated using spoor frequency and radio-telemetry for leopards and lions, respectively. Van Dyke (1986), on the other hand, using the same methods only found a weak, but significant relation for mountain lions while Vincent *et al.* (1991) and Serverson and Plumb (1998) had poor calibrations for roe deer (*Capreolous capreolous*) and prairie dog populations estimated using kilometre index and prairie dog burrows, respectively. All these studies were specifically designed for index calibration which was not the case for this study. In addition, study areas were smaller and animals were less mobile than cheetah, which allowed more accurate density determination.

Reliability of spoor tracking data

In order for an index to be considered a reliable method to monitor relative abundance, the index 95% confidence calibration equation should be narrow. Stander (1998), Walker *et al.* (2000), Funston *et al.* (2001), and Watson (2003) all reported narrow confidence intervals on their studies of leopards, mountain viscachas, leopards, wild dogs and lions, and fur seals, respectively. However, in this study, calibration confidence intervals were too wide and any predictions based on this model would not be reliable. Similar results were obtained by Servin *et al.* (1987) and Vincent *et al.* (1991) when calibrating prairie dog burrows and kilometre index as possible indices to monitor prairie dogs and roe deer, respectively. The high variability between samples and low spoor frequency could be the reasons for such wide confidence intervals.

Temporal and spatial repeatability of the technique

Transects A and B are surrounded by different vegetation and given that cheetah showed preference for mosaic habitat type (Marker 2002), one would expect significant differences in spoor frequency and densities between them. This was not the case for this study. Thus these results serve as an indication of the method's spatial repeatability and reiterate the possibility that a local population had been monitored over the years. This is supported by the size of the farm on which spoor tracking is conducted (i.e. $73km^2$) in relation to the minimal cheetah home range of $265km^2$. Thus, the farm is more likely to be part of a home range or could be an overlap area. In addition, cheetahs are reported to travel between 2 and 9 km on a daily basis (Eaton 1974; Hunter & Balme 2004). Marker (2002) reported a 2.5km daily travel distance for cheetah in the study area. Therefore, there is a high probability that animals easily crossed the two transects which are only 6km at maximum apart, resulting in no significant statistical difference in the transect frequencies and densities over time. Furthermore, it supports the significant difference between the combined spoor density estimates per area versus the radio telemetry, as the same individuals could easily have been double counted. Thus, pooling of data would result in index inflation.

Despite the insignificant statistical results, the high spoor frequency and low spoor density for transect B and the reverse situation for transect A could indicate that cheetahs are using roads on transect A as runways. This could result in inflated density estimates due to edge-effect (Greig-Smith 1983) and bias, particularly for areas highly encroached such as those for transect A. A similar conclusion was reached by Mahon (*et al.* 1998) in that there was a preferential use of roads by feral cats, red foxes and dingoes when assessing the reliability of spoor counts on roads to monitor them.

In terms of temporal sensitivity of spoor tracking to population changes, Elandsvreugde is a research farm where removals are only for research and thus the cheetah population should have remained fairly stable over the years. This explains our findings of no major annual differences in frequencies and densities measured by spoor tracking for transect B and the combined transects. The opposite is expected for farms where removals via trapping or killing occur as a preventative measure against livestock or game depredation (Marker *et al.* 2003). These removals result in an empty habitats and niche that may result temporarily in an increase in cheetah activity and scent-marking and thus spoor in the area until ownership is established (Marker 2002). This may explain the significant difference between years in both spoor frequency and density along transect A.

Relation between spoor frequency or sampling variation and sampling effort

Jackson *et al.* (2005) stated that large sample sizes are needed for index calibrations. Determinations of necessary sample effort to attain certain precision levels are common in the literature. Stander (1998) established a sample effort of 1200 km for leopards using spoor and Funston *et al.* (2001), using the same method for lions, stipulated 1100 km and 2150 km on tree and dune-savanna habitats respectively. Similarly, van Dyke *et al.* (1986), also using spoor tracking, ascertained that under ideal sampling conditions (i.e. snow) for mountain lion, less than 90 km or 500 km^2 would need to be surveyed in order to find tracks of any mountain lion in the area. This increased by 300% when tracking conditions were not ideal. These studies showed that lower sampling efforts were required than for the Namibian cheetah (1655 km), except for lions on the dune-savanna habitat. Such high sampling efforts can be expected for cheetah as they live at low densities (Caro 1994), are far ranging (Hunter & Hamman 2003) and have home ranges larger in Namibia than in other range countries (Marker 2002). Thus, discrepancies in required effort are partially a reflection of the different spatial and social organization, and partially due to different substrate structure. Nevertheless, there was consistency in the results of Stander (1998) and Funston *et al.* (2001) in that spoor frequency showed a drastic decrease at low sample effort levels.

Conclusion and Recommendations

This study attempted to evaluate and calibrate spoor density ascertained via spoor frequency against radio-telemetry densities for cheetah in Namibia. Poor regression results indicate that spoor frequency as measured was not be a reliable method to derive relative density estimates or predict absolute abundance, at the radio-telemetry sample area scale. Based on this finding, predictions or extrapolation would be unreliable. A number of reasons were proposed for such poor results. The spatial utilization and substrate quality were two main environmental constraints limiting precision whereas transect design limited the index power. Nevertheless, this study highlighted three major aspects that need consideration when spoor frequency is to be implemented as a means to monitor free ranging cheetah.

(1) A pre-requirement would be to assess the extent to which cheetahs do make use of roads in the study area by monitoring different transects. (2) Based on this, plus on other ecological information (i.e. spatial distribution, daily distances traveled, and habitat preferences), transect layout should be as straight as possible, in order to encompass different home ranges. (3) Concurrent to spoor tracking, it is recommended that local relative density estimates are obtained using different census techniques such as camera-trapping. It is recommended to have the cameras placed in close proximity to spoor transects as described by Silveira *et al.* (2003) as this would aid with spoor validation. The incorporation of rigorous spoor validation is vital as it will permit the application of other analytical methods such as mark-recapture.

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