Indexing small mammalian carnivores in the southern Kalahari, South Africa

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Abstract. Monitoring animal populations in changing environments is crucial to wildlife conservation and management, but restrictions in resources are a recurring problem for wildlife managers and researchers throughout Africa. Land-use-induced shrub encroachment in Kalahari savannah rangelands has led to fragmentation of the landscape. Mammalian carnivores are particularly vulnerable to local extinction in fragmented landscapes, but their low numbers and their often nocturnal and secretive habits make them difficult to monitor. In this study, we tested the applicability of a passive tracking method and compared two measurement methods and index calculations for monitoring small carnivores across a grazing gradient in the southern Kalahari. During the four years of monitoring in a five-year period, we used the knowledge of indigenous Khoisan Bushmen for the identification of carnivore tracks on 640 sand transects (5 m × 250 m). Our results showed that this simple and inexpensive observation method enabled detailed monitoring of 10 small carnivore species across the grazing gradient. A binary index calculated an index based on presence/absence of a species' tracks on each transect, whereas the track intrusion index used the number of track intrusions to each transect for each species in its calculations. For less common species, the two indices were similar in trend and magnitude, because the number of intrusions to each transect was typically 1 or 0. Usually, the two indices showed relatively strong correlations. However, species with patchy distributions of higher numbers presented difficulties for the binary index to monitor trends, but not for the track intrusion index.

Introduction

One of the most threatening forms of rangeland degradation in savannas worldwide is landscape changes resulting from shrub encroachment induced by heavy grazing (e.g. Archer *et al*. 1995; Jeltsch *et al*. 1998; Cabral *et al*. 2003; Sankaran *et al*. 2005). In particular, shrub encroachment has become increasingly evident over the past five decades in the savanna of the southern Kalahari (Leistner 1967; Skarpe 1991; Tews *et al*. 2006). Area-wide shrub encroachment has led to habitat fragmentation of Kalahari savanna rangelands and reduced abundance and diversity across animal taxa (birds: Thiele *et al*. in press; rodents: Blaum *et al*. 2007a; Blaum and Wichmann 2007; mammalian carnivores: Blaum *et al*. 2007b, 2007c, 2007d). While habitat fragmentation depends on the spatial mobility of the studied organisms, both mammals and birds respond mostly to large-scale changes in the environment.

Monitoring animal populations in such changing environments is crucial to wildlife conservation and management, but restrictions in resources are a recurring problem for wildlife managers and researchers throughout Africa (e.g. Engeman and Evangilista 2007). Lack of equipment and trained personnel require field methods to be simple and inexpensive, yet effective in detecting changes in wildlife densities. Furthermore, field methods that allow simultaneous monitoring of a suite of species are particularly valuable.

In fragmented landscapes, such as in Kalahari savanna rangelands, mammalian carnivores are particularly vulnerable

to local extinction (Woodroffe and Ginsberg 1998; Crooks 2002; Blaum *et al*. 2007b, 2007c, 2007d). Despite the suitability of carnivores as indicator species for conservation planning (Soule and Terborgh 1999), they are difficult to study because of their low numbers, often nocturnal and secretive habits and wariness of humans (Sargeant *et al*. 1998). Moreover, managers responsible for monitoring wildlife populations for conservation often have inadequate training, understanding, or resources to implement complex density estimations or other monitoring methods. These contexts have created the need for monitoring methodology that is rapid, easy to conduct, and records multiple species with different activity patterns, for which field personnel can employ the methods with the consistency needed for valid quantitative results.

Various passive tracking methods have been successfully used to monitor carnivores in Australia (Allen *et al*. 1996), Africa (Engeman and Evangilista 2007) and North America (Engeman *et al*. 2000, 2002, 2003a, 2003b, 2005). Hence, we considered this as a low-tech, but proven, approach for monitoring the small carnivores of the southern Kalahari. In this study, we combined passive tracking sampling with indigenous knowledge of local Khoisan Bushmen to produce methodology especially suitable to the habitat, climate and small carnivores of the southern Kalahari. For millennia the Khoisan have lived as hunters and gatherers in the Kalahari and have evolved detailed knowledge on collecting plants (e.g. devil's claw,

Harpagophytum procumbens, medicinal plant) and tracking animals (Traill 1998). Experienced Khoisan trackers readily identify to species level animal tracks of different taxonomic groups (e.g. rodents, ungulates, small and large carnivores, some birds and some insect species) (Spinney 1998; Conniff 2001; Liebenberg 2006).

We also examined two mathematical formulations for constructing indices from the track data. We compared and evaluated their applicability to the Kalahari carnivore populations. We comment on their suitability there as a general monitoring method for these carnivores, particularly in rangelands with varying degrees of shrub encroachment.

Methods

Study area

Our study area was in the southern Kalahari rangeland south of the Kgalagadi Transfrontier Park between Twee Rivieren and Askham (26°15′S, 20°35′E) in the Northern Cape Province, South Africa. The savanna vegetation is the western form of the Kalahari Thornveld, which typically consists of trees (*Acacia erioloba*, *A. haematoxylon and Boscia albitrunca*) and shrubs (*Rhigozum trichotomum* and *Acacia mellifera*) sparsely scattered in a grassy matrix (*Stipagrostis* spp., *Eragrostis* spp. and *Schmidtia kalahariensis*) (Leistner 1967). The long-term average annual rainfall of the study area is 174 mm predominantly occurring from January to April and extreme temperatures range from winter lows of –10.3°C to summer highs of up to 45.4°C (Van Rooyen *et al*. 1990).

We selected five farms with mixed grazing of cattle and sheep where shrub cover ranged from low (<5%) to high (>25%) (Farm 1, 4%; Farm 2, 9%; Farm 3, 15%; Farm 4, 18%; Farm 5, 26%). The farms differed in their historic and present grazing impact, and current stocking rates (during the last 10 years) ranged from 6 livestock units per 100 ha on the heavily shrub encroached farm to 2 livestock units per 100 ha on the farm without shrub encroachment. Despite the different grazing impact, all study sites were typical for the Kalahari Thornveld. The maximum distance between farms was 40 km. The distance between the two closest farms was 7.5 km, which is large in comparison to the home-range sizes of the observed carnivore species (Skinner and Chimimba 2005 and references therein). Therefore, we could safely assume the farms represented independent samplings of our carnivore species.

Passive carnivore survey

Carnivore tracks were monitored during the rainy season (February–May) from 2001 to 2003 and 2005. For each farm, we established four subplots (1 km × 2.5 km) with a minimum inter-subplot spacing of 2.0 km. For each subplot we established 8 transects (5 m × 250 m) on sand along dune crests where vegetation cover was lower than 5% (mainly *Stipagrostis amabilis*) with a minimum intertransect spacing of 625 m (4 years × 5 farms × 4 subplots × 8 transects). We did not need to manually prepare the sand surface because almost daily strong winds (between 12:00 and 14:00 hours) cleared old tracks. Surveys were passive (e.g. Allen *et al*. 1996) in that no attractants or other means were used to draw carnivores to the transects. Transects were monitored for 2 h, beginning 1 h after sunrise. Therefore, we were able to observe the small carnivore tracks deposited within the previous 18 h for each transect. The data recorded for each transect were the number of track intrusions by each carnivore species. Andrew Kruiper, a Khoisan Bushman tracker from the Kgalagadi Transfrontier Park identified the tracks.

Indexing carnivore abundance on the basis of presence/absence of tracks

We first calculated an index based on binary (presence/absence) data from each transect, which we refer to as BI (for binary index). For each year and farm we calculated the proportion of transects with tracks (P) per species within each subplot on each farm (modified after Conner *et al*. 1983). For a given species, the value of P for the jth subplot is

$$P_j = n_j / t_j$$

where n_j is the number of transects in subplot j with at least one track intrusion for that species, and t_j is the number of transects in subplot j. The BI value for that species on a given farm is the mean of the P_j values over the four subplots.

$$BI = (\Sigma P_j)/4$$

Note that if the same number of transects were observed in each subplot (i.e. $t_j = t$, a constant), then.

$$BI = (\Sigma n_j)/4t.$$

Therefore, except in unusual circumstances, $t_j = 8$, and, hence,

$$BI = (\Sigma n_j)/32.$$

Given the minimum 2.0-km distance between subplots, we assume their independence for our species of interest. Therefore.

$$Var(BI) = \Sigma Var(P_j)/16,$$

where $Var(P_j)$ is estimated as $(P_j)(1 - P_j)/t_j = (P_j)(1 - P_j)/8$.

Indexing carnivore abundance based on the number of track intrusions

Using the number of sets of tracks (number of intrusions by each species into a transect) allows the calculation of an index along the lines of the passive tracking index (PTI) successfully used to monitor carnivore abundance (and many other species: Allen *et al*. 1996; Engeman *et al*. 2000, 2001, 2002; Engeman and Evangilista 2007). We calculated a track intrusion index (TII) and associated variance analogous to the general paradigm of Engeman (2005). A linear random-effects model is used to describe the number of intrusions onto each transect in each subplot, but the term for time (days) given by Engeman (2005) is replaced by a term for subplot. The mean number of track intrusions on each transect by each species is calculated for each subplot. The index values are the means of the subplot means for each species (modified after Engeman 2005):

$$TII = \frac{1}{s} \sum_{j=1}^{s} \frac{1}{t_j} \sum_{i=1}^{t_j} x_{ij}$$

where the x_{ij} represent the number of intrusions by a given species on transect i on subplot j, s is the number of subplots, and t_j is the number of transects contributing data on subplot j.

Since eight transects were observed in each of the four subplots, the TII formula when no data are missing simplifies to:

$$TII = \frac{1}{32} \sum_{j=1}^{4} \sum_{i=1}^{8} x_{ij},$$

SAS PROC VARCOMP with a restricted maximum-likelihood estimation procedure (REML) (SAS Institute 2004) was used to calculate the variance components (Searle *et al.* 1992) needed in the TII variance estimation formula (Engeman 2005):

$$\text{var}(TII) = \frac{\sigma_t^2}{s} \sum_{j=1}^{s} \frac{1}{t_j} + \frac{\sigma_s^2}{s} + \frac{\sigma_e^2}{s^2} \sum_{j=1}^{s} \frac{1}{t_j},$$

where σ_t^2, σ_s^2, and σ_e^2 are, respectively, the components for between-transect variability, subplot variability and random observational variability associated with each farm. Additional simplification is achieved if no data are missing (all transects in each subplot contribute data):

$$\text{var}(TII) = \frac{\sigma_t^2}{4} \sum_{j=1}^{s} \frac{1}{8} + \frac{\sigma_s^2}{4} + \frac{\sigma_e^2}{16} \sum_{j=1}^{s} \frac{1}{8}$$

$$= \frac{\sigma_t^2}{8} + \frac{\sigma_s^2}{4} + \frac{\sigma_e^2}{32}$$

Comparison of index formulations

Although the two calculation methods are, to a degree, necessarily correlated because they are transformations of the same raw data, we calculate correlations between the methods for each species as a descriptive means for indicating degree of correspondence. Since our ultimate purpose for monitoring small carnivores was to use them as indicators of habitat degradation on wildlife, we also examined the relationships of the indices within the gradient of habitats represented by the farms, with Farm 1 having the lowest shrub cover (4%) and Farm 5 being the most degraded by shrub encroachment (25.5%). We expected the greatest differences in inferences between the two index methods to occur for situations where populations were high, because that would be where the most information would be lost from reducing continuous data to binary data (e.g. Engeman *et al.* 1989, 2000; Allen *et al.* 1996). Where populations are very low, the transects would serve primarily as a detection method, and would likely result in counts of track intrusions generally appearing binary. Examining the correlations between the

indices for each species on a farm-by-farm basis would more readily isolate different population levels within each species to assess correspondence between the indices.

Results

We indexed 10 species of small carnivores on the five farms (Table 1). Each species was found on each farm, although not all species were found on each farm each year (Tables 2, 3). The species with the most commonly observed tracks were the yellow mongoose, African wild cat, and striped polecat, while the least commonly observed species were the small-spotted cat, slender mongoose and suricate (also known as the meerkat) (Tables 2, 3). Values for both indices were highest for Farm 1 and lowest for Farm 5 for almost every species and year, and also tended to follow the gradient of habitat degradation by shrub encroachment for the farms in between (Tables 2, 3). Most of the standard errors of TII (across species and years) were higher than of BI.

The hazards of attempting to infer trends with binary data for populations that occur in patchy abundance is well illustrated with the results for the yellow mongoose on Farm 4 (Table 3). The BI indicated a steady, if not increasing, population trend from 2001 to 2005, with values of 0.38, 0.38, 0.44, 0.44. However, the TII indicated a decreasing trend, with values of 1.06, 0.81, 0.44, 0.44. The last two survey years (2003 and 2005) yielded, at most, one track intrusion per transect, resulting in the equality between the BI and TII values. In these years, standard errors of TII and BI were very similar or even identical (e.g. standard errors of TII and BI for slender mongoose on Farms 2 and 5 in 2005 were 0.03). However, in the first two years there were substantially more track intrusions per transect, but fewer transects with tracks. That is, if a track intrusion occurred on a transect, then there were likely to be other track intrusions on the same transect, even though fewer transects may have received tracks than in later years. This resulted in higher values for TII, but lower values for BI. This opposite trend produced a correlation of –0.94 (Table 4). In general, the correlation values in Table 4 are instructive. Further examples of how the TII is better able to elucidate trends for more numerous species is illustrated in Fig. 1 showing four species on Farm 1. The TII describes severe fluctuations in the populations of African wildcat and striped polecat, but the BI cannot. For the less common cape fox and

Table 1. The common and scientific names and habit descriptions (after Skinner and Chimimba 2005) for the species of small carnivores tracked on five farms in the Kalahari Thornveld in the Northern Cape Province, South Africa

Common name	Scientific name	Habits (activity; foraging)
Yellow mongoose (YM)	*Cynictis penicillata*	Diurnal; solitary
African wild cat (AWC)	*Felis lybica*	Nocturnal; solitary
Striped polecat (SP)	*Ictonyx striatus*	Nocturnal, solitary
Bat-eared fox (BEF)	*Octocyon megalotis*	Diurnal in winter, nocturnal in summer; groups
Cape fox (CF)	*Vulpes chama*	Nocturnal; solitary
Small-spotted Genet (SSG)	*Genetta genetta*	Nocturnal; solitary
Black-backed jackal (BBJ)	*Canis mesomelas*	Nocturnal and diurnal; pairs/ groups
Small-spotted cat (SSC)	*Felis nigripes*	Nocturnal; solitary
Slender mongoose (SM)	*Galerella sanguinea*	Diurnal; solitary
Suricate (SU)	*Suricata suricata*	Diurnal; social

Table 2. Index values for the transect intrusion index (TII ± s.e.) for 10 carnivore species from five farms over four years in the Kalahari Thornveld in the Northern Cape Province, South Africa

For explanation of species labels see Table 1

Year	Farm	YM	AW	SP	BEF	CF	SSG	BBJ	SSC	SM	SU
2001	1	1.22 ± 0.26	1.38 ± 0.36	1.50 ± 0.34	0.22 ± 0.12	0.50 ± 0.15	0.22 ± 0.15	0.34 ± 0.24	0.00 ± 0.00	0.09 ± 0.09	0.25 ± 0.25
	2	1.63 ± 0.17	0.88 ± 0.21	1.34 ± 0.30	0.31 ± 0.17	0.31 ± 0.10	0.44 ± 0.19	0.00 ± 0.00	0.06 ± 0.06	0.06 ± 0.06	0.84 ± 0.59
	3	1.56 ± 0.29	0.44 ± 0.12	1.03 ± 0.29	0.19 ± 0.19	0.31 ± 0.16	0.31 ± 0.23	0.31 ± 0.22	0.09 ± 0.09	0.09 ± 0.09	1.19 ± 0.73
	4	1.06 ± 0.38	0.38 ± 0.14	0.31 ± 0.13	0.38 ± 0.27	0.28 ± 0.21	0.44 ± 0.17	0.22 ± 0.13	0.00 ± 0.00	0.00 ± 0.00	0.66 ± 0.46
	5	0.25 ± 0.10	0.25 ± 0.15	0.03 ± 0.03	0.09 ± 0.07	0.09 ± 0.06	0.16 ± 0.09	0.06 ± 0.04	0.00 ± 0.00	0.03 ± 0.03	0.19 ± 0.19
2002	1	1.31 ± 0.33	0.72 ± 0.24	0.91 ± 0.37	0.16 ± 0.09	0.56 ± 0.20	0.22 ± 0.22	0.34 ± 0.34	0.03 ± 0.03	0.06 ± 0.06	0.15 ± 0.16
	2	1.44 ± 0.22	0.66 ± 0.36	0.72 ± 0.33	0.19 ± 0.09	0.19 ± 0.11	0.09 ± 0.09	0.16 ± 0.16	0.00 ± 0.00	0.03 ± 0.03	0.59 ± 0.59
	3	1.22 ± 0.20	0.50 ± 0.09	0.75 ± 0.29	0.34 ± 0.18	0.25 ± 0.21	0.38 ± 0.16	0.09 ± 0.09	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
	4	0.81 ± 0.21	0.31 ± 0.09	0.19 ± 0.07	0.19 ± 0.12	0.28 ± 0.15	0.38 ± 0.22	0.03 ± 0.03	0.00 ± 0.00	0.03 ± 0.03	0.41 ± 0.41
	5	0.22 ± 0.09	0.41 ± 0.18	0.19 ± 0.08	0.06 ± 0.06	0.09 ± 0.07	0.06 ± 0.06	0.03 ± 0.03	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
2003	1	0.53 ± 0.11	0.38 ± 0.09	0.38 ± 0.09	0.06 ± 0.06	0.53 ± 0.27	0.22 ± 0.13	0.28 ± 0.20	0.00 ± 0.00	0.06 ± 0.06	0.38 ± 0.20
	2	0.91 ± 0.18	0.84 ± 0.23	0.84 ± 0.20	0.09 ± 0.07	0.19 ± 0.08	0.25 ± 0.11	0.13 ± 0.13	0.06 ± 0.06	0.03 ± 0.03	0.00 ± 0.00
	3	0.59 ± 0.12	0.34 ± 0.12	0.75 ± 0.23	0.31 ± 0.17	0.25 ± 0.08	0.16 ± 0.11	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
	4	0.44 ± 0.09	0.28 ± 0.10	0.13 ± 0.06	0.16 ± 0.11	0.03 ± 0.03	0.06 ± 0.06	0.03 ± 0.03	0.03 ± 0.03	0.03 ± 0.03	0.00 ± 0.00
	5	0.22 ± 0.10	0.22 ± 0.10	0.16 ± 0.08	0.03 ± 0.03	0.09 ± 0.08	0.09 ± 0.07	0.06 ± 0.06	0.00 ± 0.00	0.03 ± 0.03	0.22 ± 0.22
2005	1	0.94 ± 0.16	1.00 ± 0.22	1.03 ± 0.31	0.03 ± 0.03	0.25 ± 0.15	0.09 ± 0.09	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
	2	0.97 ± 0.17	1.03 ± 0.27	0.78 ± 0.18	0.16 ± 0.12	0.16 ± 0.08	0.25 ± 0.08	0.13 ± 0.10	0.03 ± 0.03	0.03 ± 0.03	0.25 ± 0.25
	3	0.84 ± 0.23	0.44 ± 0.11	0.81 ± 0.19	0.19 ± 0.12	0.41 ± 0.33	0.41 ± 0.26	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
	4	0.44 ± 0.15	0.31 ± 0.08	0.09 ± 0.05	0.09 ± 0.09	0.06 ± 0.06	0.06 ± 0.06	0.00 ± 0.00	0.06 ± 0.06	0.06 ± 0.06	0.00 ± 0.00
	5	0.16 ± 0.07	0.13 ± 0.06	0.09 ± 0.05	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.22 ± 0.22

small-spotted genet, the two methods are more similar in trend. Over all farms and years, BI and TII shows high correlation coefficients for most species. The low correlation for slender mongoose is somewhat deceptive in that correlations could not be calculated for three of the five farms since all zero values produced zero variances. Nevertheless, on a within-farm basis, as seen above, the two indices sometimes showed conflicting trends over time (Table 4).

Both indices were sensitive in detecting the effects of habitat degradation (here amount of shrub cover) on carnivore abundance. For example, relative abundance of African wildcats and cape foxes declined with shrub cover (Fig. 2). TII and BI showed similar significant trends for the less common cape fox, which would be expected since the count data were essentially binary. However, the TII more sensitively described the decrease for the common African wildcat. The abundance of all carnivores was

Table 3. Index values for the binary index (BI ± s.e.) for 10 carnivore species from five farms over four years in the Kalahari Thornveld in the Northern Cape Province, South Africa

For explanation of species labels see Table 1

Year	Farm	YM	AW	SP	BEF	CF	SSG	BBJ	SSC	SM	SU
2001	1	0.53 ± 0.09	0.47 ± 0.08	0.56 ± 0.09	0.09 ± 0.05	0.31 ± 0.07	0.09 ± 0.05	0.06 ± 0.04	0.00 ± 0.00	0.03 ± 0.03	0.03 ± 0.03
	2	0.84 ± 0.06	0.47 ± 0.08	0.69 ± 0.07	0.19 ± 0.06	0.28 ± 0.08	0.28 ± 0.08	0.00 ± 0.00	0.03 ± 0.03	0.03 ± 0.03	0.06 ± 0.04
	3	0.53 ± 0.09	0.38 ± 0.08	0.38 ± 0.09	0.06 ± 0.04	0.19 ± 0.06	0.13 ± 0.05	0.06 ± 0.04	0.03 ± 0.03	0.03 ± 0.03	0.09 ± 0.05
	4	0.38 ± 0.08	0.38 ± 0.08	0.22 ± 0.07	0.13 ± 0.05	0.13 ± 0.05	0.19 ± 0.06	0.09 ± 0.05	0.00 ± 0.00	0.00 ± 0.00	0.06 ± 0.04
	5	0.19 ± 0.07	0.16 ± 0.06	0.03 ± 0.03	0.06 ± 0.04	0.09 ± 0.05	0.09 ± 0.05	0.06 ± 0.04	0.00 ± 0.00	0.03 ± 0.03	0.03 ± 0.03
2002	1	0.66 ± 0.08	0.44 ± 0.07	0.44 ± 0.07	0.09 ± 0.05	0.22 ± 0.07	0.13 ± 0.04	0.06 ± 0.04	0.03 ± 0.03	0.03 ± 0.03	0.03 ± 0.03
	2	0.72 ± 0.08	0.31 ± 0.06	0.31 ± 0.06	0.13 ± 0.06	0.16 ± 0.06	0.06 ± 0.04	0.03 ± 0.03	0.00 ± 0.00	0.03 ± 0.03	0.06 ± 0.04
	3	0.63 ± 0.08	0.50 ± 0.08	0.50 ± 0.08	0.13 ± 0.06	0.09 ± 0.05	0.22 ± 0.07	0.03 ± 0.03	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
	4	0.38 ± 0.08	0.31 ± 0.08	0.19 ± 0.07	0.09 ± 0.05	0.13 ± 0.06	0.16 ± 0.06	0.03 ± 0.03	0.00 ± 0.00	0.03 ± 0.03	0.03 ± 0.03
	5	0.22 ± 0.07	0.19 ± 0.07	0.19 ± 0.07	0.06 ± 0.04	0.06 ± 0.04	0.03 ± 0.03	0.03 ± 0.03	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
2003	1	0.47 ± 0.09	0.38 ± 0.08	0.38 ± 0.08	0.03 ± 0.03	0.19 ± 0.07	0.09 ± 0.05	0.06 ± 0.04	0.00 ± 0.00	0.03 ± 0.03	0.03 ± 0.03
	2	0.69 ± 0.08	0.50 ± 0.09	0.50 ± 0.09	0.06 ± 0.04	0.19 ± 0.06	0.16 ± 0.06	0.03 ± 0.03	0.03 ± 0.03	0.03 ± 0.03	0.00 ± 0.00
	3	0.59 ± 0.08	0.34 ± 0.08	0.34 ± 0.08	0.19 ± 0.06	0.25 ± 0.08	0.06 ± 0.04	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
	4	0.44 ± 0.09	0.28 ± 0.08	0.13 ± 0.06	0.06 ± 0.04	0.03 ± 0.03	0.03 ± 0.03	0.03 ± 0.03	0.03 ± 0.03	0.03 ± 0.03	0.00 ± 0.00
	5	0.22 ± 0.07	0.16 ± 0.06	0.13 ± 0.06	0.03 ± 0.03	0.09 ± 0.05	0.06 ± 0.04	0.06 ± 0.04	0.00 ± 0.00	0.03 ± 0.03	0.03 ± 0.03
2005	1	0.63 ± 0.08	0.44 ± 0.08	0.44 ± 0.08	0.03 ± 0.03	0.09 ± 0.05	0.03 ± 0.03	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00
	2	0.75 ± 0.08	0.44 ± 0.09	0.44 ± 0.09	0.16 ± 0.06	0.16 ± 0.06	0.25 ± 0.07	0.06 ± 0.04	0.03 ± 0.03	0.03 ± 0.03	0.03 ± 0.03
	3	0.66 ± 0.08	0.44 ± 0.09	0.44 ± 0.09	0.19 ± 0.06	0.19 ± 0.05	0.25 ± 0.06	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
	4	0.44 ± 0.07	0.31 ± 0.08	0.09 ± 0.05	0.03 ± 0.03	0.06 ± 0.04	0.03 ± 0.03	0.00 ± 0.00	0.06 ± 0.04	0.06 ± 0.04	0.00 ± 0.00
	5	0.16 ± 0.06	0.13 ± 0.06	0.09 ± 0.05	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.03 ± 0.03

Table 4.   Correlation coefficients between the transect intrusion index (TII) and binary index (BI) overall and within five farms over four years in the Kalahari Thornveld in the Northern Cape Province, South Africa
Blank cells indicate where a zero variance was found in the denominator of the correlation calculations

Species	Overall	Farm 1	Farm 2	Farm 3	Farm 4	Farm 5
Yellow mongoose	0.78	0.67	0.71	−0.54	−0.94	0.62
African wild cat	0.70	0.93	0.66	0.90	1.00	0.98
Striped polecat	0.92	0.96	0.94	−0.30	0.94	0.98
Bat-eared fox	0.79	0.95	0.87	0.26	0.96	0.94
Cape fox	0.76	0.74	0.97	0.15	0.99	0.94
Small-spotted genet	0.90	0.89	0.89	0.97	1.00	0.99
Black-backed jackal	0.75	0.98	0.74	0.98	0.97	1.00
Small-spotted cat	0.86	1.00	0.89	1.00	1.00	–
Slender mongoose	0.55	–	–	0.66	1.00	–
Suricate	0.96	0.81	0.95	1.00	0.99	0.99

low at Farm 5 with the highest degree of habitat degradation but showed different responses to low and average grazing impact on the other farms (Table 3).

Discussion

In this study, we tested the applicability of a passive tracking method and compared two index measurement and calculation methods for monitoring small carnivore populations in southern Kalahari rangelands. Our results clearly showed that this simple and inexpensive passive tracking method could detect and allow

monitoring of the relative abundances of 10 small carnivore species across a grazing gradient on five farms.

Worldwide, carnivore populations are directly affected by hunting and poaching (e.g. Treves and Karanth 2003) but also indirectly by land-use-induced habitat loss and fragmentation (Woodroffe and Ginsberg 1998; Crooks 2002; Blaum *et al.* 2007*b*, 2007*c*, 2007*d*). In southern Kalahari rangelands, hunting pressure on small carnivores is low, although this is not true for black-backed jackals because of their negative economic impact through depredation on domestic livestock (e.g. sheep, lambs). In contrast, land-use-induced shrub encroachment that substantially changes savannah habitats greatly reduces local carnivore densities (Blaum *et al.* 2007*b*, 2007*c*, 2007*d*; Popp *et al.* 2007).

Region-wide monitoring of carnivore populations in Kalahari rangelands is a great challenge for conserving and managing wildlife outside protected areas. This is particularly the case for the southern Kalahari, where the size of a single farm varies from 3000 ha to more than 30 000 ha in a mosaic of well over 500 farms. Our results demonstrated (1) the sensitivity of the passive tracking observation method to provide suitable data to detect carnivores and monitor changes in their abundances, (2) the relationships between the two index-calculation methods and a potential pitfall for using binary data, (3) the potential for a cost-efficient, region-wide monitoring method and (4) the use of indigenous Khoisan knowledge for tracking carnivores.

Monitoring of the small Kalahari carnivores is particularly difficult because most species are nocturnal or otherwise cryptic,
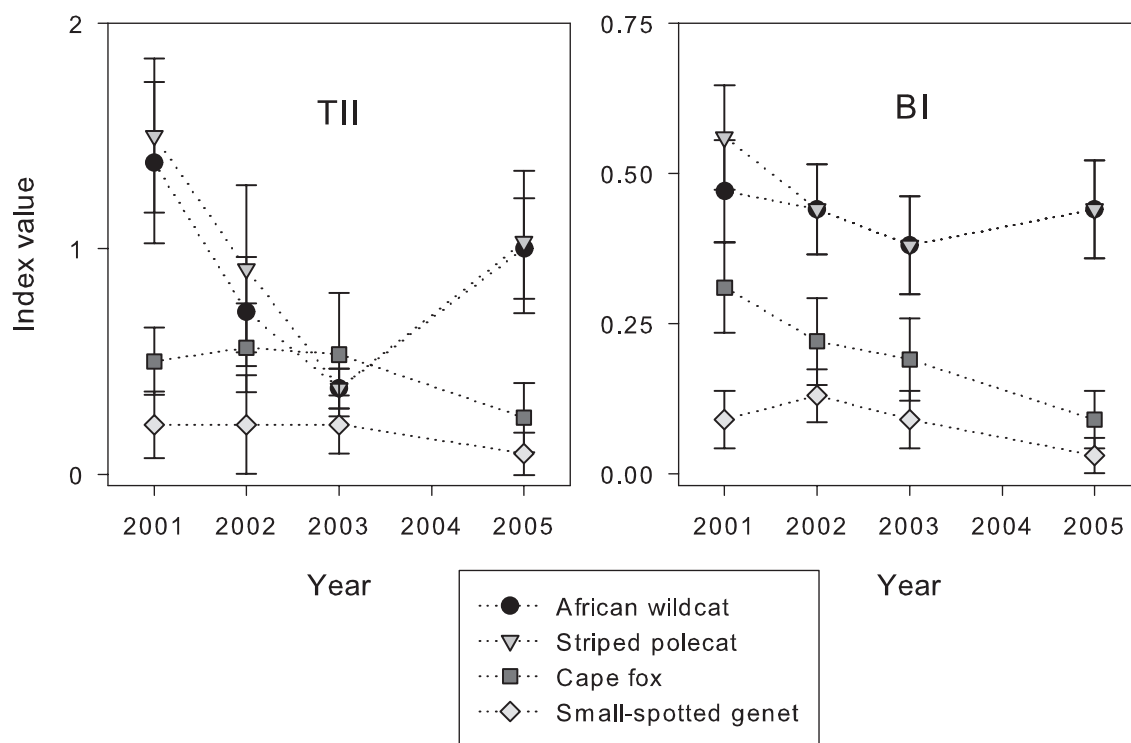

Fig. 1.   Trends in index values for the transect intrusion index (TII) and binary index (BI) for study Farm 1 (shrub cover = 4%) of the Kalahari Thornveld in the Northern Cape Province, South Africa.
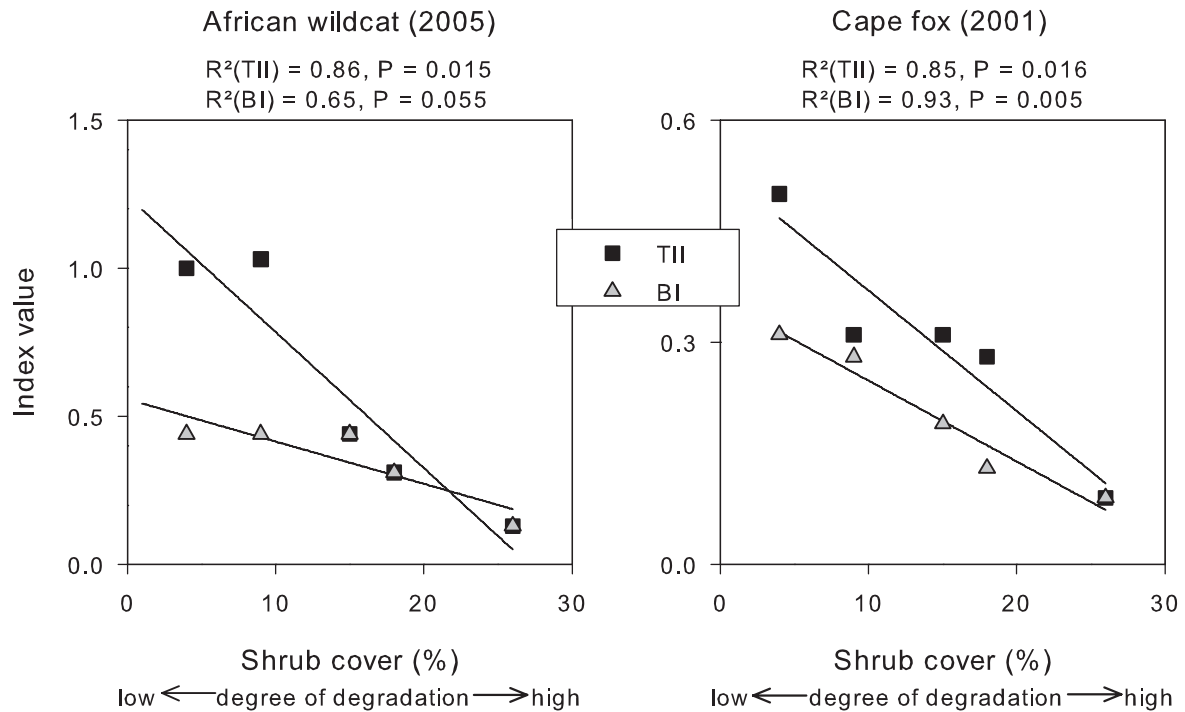
African wildcat (2005)

R²(TII) = 0.86, P = 0.015
R²(BI) = 0.65, P = 0.055

Cape fox (2001)

R²(TII) = 0.85, P = 0.016
R²(BI) = 0.93, P = 0.005

Fig. 2. Relationship between habitat degradation (shrub cover) and abundance index value (African wildcat and cape fox) on five Kalahari farms in the Northern Cape Province, South Africa. Black squares show index calculations of the transect intrusion index (TII) and grey triangles calculations of the binary index (BI).

and knowledge of their ecology is scant, except for the diurnal suricates (e.g. Doolan and MacDonald 1996; Clutton-Brock *et al*. 1999) and yellow mongooses (e.g. Earlé 1981; Rasa *et al*. 1992; Blaum *et al*. 2007*b*). Although passive tracking indices cannot measure absolute carnivore densities, they can reflect the relative abundances of the species across farms (Conner *et al*. 1983; Sargeant Johnson and Berg 1998; Crooks 2002; Engeman 2005).

The calculations for the two indices were based on two different measurements. The BI was calculated with binary data (presence/absence of a species) on a tracking transect, whereas the number of track intrusions per transect were used for TII. Thus, as expected, variances calculations of TII were higher than of BI, and were similar, or identical, when the number of track intrusions of a species per transect approximated 1. While the correlations between BI and TII were high for most species (Table 4), our results showed that the two methods may indicate differing population trends, especially if the animals occur in spatially patchy population patterns of higher abundance, as was seen by the decreasing versus increasing trends shown for the yellow mongoose on Farm 4 for the BI and TII, respectively (Fig. 1). This suggests a higher sensitivity of TII for detecting changes in population trends of more numerous species, especially when the populations are not distributed uniformly over the area of interest. One might also argue that the multiple intrusions occurring on a single transect could have been made by the same animal, in which case the BI index is less biased. Nevertheless, although most small carnivores are solitary foragers, they move in family groups when raising the young (Skinner and Chimimba 2005). As we have monitored the carnivores during this important time of reproduction, TII records

track intrusions of adults that are accompanied by their offspring. Indeed, at the same section of a transect we frequently observed tracks of 1–2 large individuals next to tracks of 1–3 remarkably smaller individuals. Therefore, we explain the decreasing TII for the yellow mongoose on Farm 4 by a reduction in reproductive success from 2001 to 2005 while the number of adult individuals changed only marginally.

These differences in sensitivity between TII and BI are further supported by the relationship between habitat degradation (amount of shrub cover) and the abundance of African wildcats and cape foxes (Fig. 2). While both indices suggest a similar significant decrease in abundance of the less common cape fox with shrub cover (degree of habitat degradation), the TII was far more sensitive to the relationship for the African wildcat (Fig. 2). For the cape fox, the slopes of the regression lines of TII and BI are similar (because the track count data were essentially binary), but were very different for the African wildcat.

The responses of carnivores to habitat degradation varied between species. For example, we found a linear relationship between shrub cover and African wildcats whereas our results suggest a hump-shaped relationship for small spotted genets and bat-eared foxes (Tables 2, 3).

Nevertheless, continuous monitoring is necessary to show whether, for example, the reduction in reproductive success remains and results in a decline in population size, or whether neighbouring individuals immigrate and sustain a constant local population. Our results suggest that TII can also monitor changes in reproductive success of solitary species and can probably be used as an indicator for nearby future population trends.

Historically, most survey methods relying on tracking plot data the world over have recorded binary data (e.g. Engeman *et al.* 2000; Crooks 2002; Engeman 2005). A variety of recent studies have demonstrated superior sensitivity using counts of multiple track intrusions compared with presence/absence data (e.g. Allen *et al.* 1996; Engeman *et al.* 2000, 2001, 2003*b*). On the other hand, when applying strip transects, as we have done, the use of the BI can reduce labour costs when only one or two species are monitored. For binary measurements, observations on a transect can be halted immediately when a track of the focal species has been recorded, because any further tracks would not change BI. Thus, BI might be considered when funds for labour are limited, only one or two species monitored, and/or few observations of the target species are expected (that is, counts of the number of track intrusions will typically be 0 or 1). Also in our study the number of track intrusions of a species was sometimes 1, which resulted in identical index and variance values for BI and TII. Nevertheless, our results clearly show TII to be more sensitive for detecting population differences.

Acknowledgements

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