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A participatory counting method to monitor populations of large mammals in non-protected areas: a case study of bicycle counts in the Zambezi Valley, Zimbabwe

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Abstract. The sustainable use of wildlife resources within community-based programmes is considered as a valuable option to enhance development and conservation objectives. The application of such a concept in communal lands requires the strong support of local communities through their active involvement in wildlife management. Precise and regular information on wildlife abundance is also essential for effective conservation. In this paper, we present an innovative participatory monitoring method based on bicycle counts developed within the framework of an integrated conservation and development project in the Zambezi Valley, Zimbabwe. Results from the 1999 count of population density and structure of common species as well as diversity of large to medium size species are presented. We demonstrate the efficiency of this method, which allowed a high census intensity with efficient animal detection, and hence appeared appropriate for use in a non-protected area. The method gave high levels of precision for the density estimates obtained (10–30% cv) and is therefore useful as a repeatable monitoring tool. In addition, this method is inexpensive to run and is easy to implement for local people. We emphasise the technical and financial autonomy offered by the bicycle counts for communities to monitor their wildlife resources, and we discuss the contribution of this method to the wildlife management process.

Introduction

Non-protected lands are of prime importance for wildlife conservation activities, since they have a crucial place within the ecological network by the significance of the area they cover and their role in the connectivity between protected areas (Bennett 1998). The sustainable management of these areas is therefore considered as a central aspect for wildlife conservation policies (Western 1989; Child and Child 1991; Halladay and Gilmour 1995). Under the classical integrated conservation and development strategy (Jacobs et al. 1987; Wells and Brandon 1992), the profitable exploitation of wildlife resources by local residents is considered as a valuable land use option to reconcile development and conservation objectives in rural areas (Hudson et al. 1989; Child and Child 1991). However, the application of such a concept in communal lands outside protected areas has to meet two main requirements to ensure a sustainable implementation. First, it must rely on the support of local communities, through their active involvement in wildlife management operations and hence decision making (KWFT (Kenya Wildlife Fund Trustees) and UNEP (United Nations Environment Programme) 1988; Kiss 1990; IIED (International Institute for Environment and Development) 1994; Hackel 1999; Hulme and Taylor 2000). Second, it requires precise and regular information on wildlife abundance and trends to ensure that management schemes are adaptive and allow for a sustainable use of wildlife populations (Kremen et al. 1994; du Toit 2002). Consequently, the implementation of a community-based conservation programme needs the development of a monitoring method that respects a compromise between its applicability by local communities, both in terms of finance and skills, and the reliability of results.

This study is part of the Biodiversity Project (Biodiversity conservation with sustainable development in the Zambezi Valley after eradication of the Tsetse fly), which was implemented in 1996 in the Dande Communal Area, Zimbabwe, to develop the sustainable use of natural resources for the benefit of local communities (Biodiversity Project 2002). This project is locally integrated through its support of the Zimbabwean CAMPFIRE programme (Communal Area Management Programme For Indigenous Resources), one of the early programmes to have promoted the actual involvement and empowerment of rural communities in the management of wildlife resources (Martin 1986; Murindagomo 1989). One of the objectives of the Biodiversity Project was to implement a system involving residents as active agents to monitor changes in large mammal populations, in order to provide information on which to make management decisions, especially on the crucial issue of quota-setting for sport hunting.

Most of the standard methodologies for assessing large African mammal densities such as aerial or car counts (Norton-Griffiths 1978) rely on technical and financial resources that are considered inappropriate for an actual local participation (Hulme and Taylor 2000). The involvement of rural residents in monitoring should build upon local skills and must be affordable to the communities' financial budget. To date, walking counts, patrols, hunting encounter rates and interviews have been proposed as methods of engaging local communities in game monitoring (IIED (International Institute for Environment and Development) 1994; Marks 1994; Matzke 1995; World Wide Fund for Nature (WWF) 1998; Noss 1999; Hulme and Taylor 2000). However, these non-standard methodologies designed to enhance participation had to be adapted to a local context and site-specific conditions, and consequently did not always meet requirements of rigour and repeatability (Hulme and Taylor 2000). In a communal area, where the encounter rate of wildlife is often low because of low animal density and high human disturbance, the efficiency of these counting methods may be questionable. The census intensity must be maximised in non-protected areas in order to increase the number of animals observed (i.e. sample size) and therefore the precision of density estimates (Kraft et al. 1995; Walsh and White 1999; Plumptre 2000). Since the origin of line-transect methods it has been emphasised that the general nature of line-transect sampling permits many methods of searching other than on foot, which can be more advantageous in increasing the distance covered and sample size, with respect to fundamental assumptions (Anderson et al. 1979).

In this paper, we present an innovative participatory monitoring technique based on bicycle counts that followed the principles of the line-transect sampling method (Anderson et al. 1979). This technique fulfils both the requirements of an active local participation and a high precision of results. In order to establish a self-sufficient technical unit, some local agents were trained to collect data, and the results from the 1999 survey are presented. A line-transect analysis was used to estimate population density for the 10 species commonly seen. Population structures of these species are also presented, as complementary information usable for wildlife management. The validity of the bicycle counting method is discussed on the basis of its efficiency for data collection as well as on its precision of population abundance estimates. We then discuss the applicability of this approach as a wildlife management tool and its suitability for the actual participation of local communities in the decision making process.

Methods

Study area

The study area is located in the middle Zambezi Valley, in Zimbabwe, between 30°00 and 31°00 longitude East and 15°30 and 16°20 latitude South. It is Communal land, constituting three Wards (2, 3 and 4) of the rural Guruve District, in the Dande Communal Area. The area is characterised mainly by former floodplains of the Zambezi river basin, at an altitude of ca. 400 m, and drained by three main rivers. It has a dry tropical climate, with low and very variable annual rainfall (on average 450–650 mm/year) and a mean annual temperature of 25 °C. Two seasons are clearly defined: a rainy season from December to March and a long dry season from April to November.

People and wildlife coexist in this communal area of 2044 km^2, which is characterised by two contrasting habitats: a dense human settlement with crop lands, and a wooded savannah. A total of 13 000 inhabitants live in this area, mainly settled along main rivers, where farming is their dominant activity (predominantly cotton and maize; Biodiversity Project 2002). Livestock populations are relatively low and localised around settlements, and although cattle numbers have been increasing recently, overgrazing does not, as yet, appear to be a problem. The uninhabited areas still cover a large proportion of the valley (>87% of the study area), and contain a remarkable species richness, with more than 40 large mammal, 200 bird and 700 plant species (Biodiversity Project 2002). The natural land cover is a deciduous dry savannah, dominated by Mopane trees (*Colophospermum mopane*) mainly associated with *Combretum apiculatum*, *C. mossambicense*, *Commiphora* spp., *Dalbergia melandoxylon*, *Diospyros kirkii*, *Kirkia accuminata*, *Sclerocarya birrea*, *Terminalia brachystemma*, *T. stuhlmannii*, *T. stenostachya, and T. sericea*. The composition

Table 1. Sample survey description: sampling intensity on transects (length, number of replicates, time of observation), with their respective number of sightings recorded, and their dominant vegetation types DVT (De Wispelaere 2001): A = dry forest, B = Jesse and Gonono bush, C = old fallows and thickets, D = dense mixed mopane woodland, E = tall mopane woodland, F = mixed mopane shrubland, G = mosaic of miombo–mopane woodland and mopane shrubbed woodland, and H = sparse vegetation on sandstone outcrops.

Transect	Length (km)	Replicates	Time (h)	DVT	Stratum area (km^2)	Sightings
1	11.4	24	81.6	A, D	231	108
2	22.0	24	59.4	F, G, H	986	120
3	3.8	29	88.3	G, H	441	159
4	10.6	30	50.5	D, F	727	133
5	9.4	47	66.0	D, F	727	173
6	14.1	17	27.3	E, F	877	42
7	3.7	22	24.8	D, F	727	112
8	23.0	35	91.1	B, E, F	968	175
9	10.4	40	96.7	C, D, E, F	1088	154
10	12.9	36	83.3	E, F	877	152
Total	121.3	304	669.0			1328

and structure of each vegetation type varies with the type of soils and forms a mosaic of woodland and shrubland varying from 4 to 18 m in height.

Data collection

Sampling design

The sample survey followed the line-transect method. The area was covered by 10 transects, between 3.7 and 23.0 km long, covering a total length of 121.3 km (Table 1). The transect lines followed established four-wheel-drive roads opened up by the Regional Tsetse and Trypanosomosis Control Program. This network was designed to systematically cover the whole area for controlling and maintaining tsetse fly targets, regardless of human activities or vegetation units. These roads were thus considered to provide a representative sample of the area. The transects start and end points were marked with red paint on trees. They were spread over the entire site, in low human disturbance areas, and chosen to allow an understandable and practical implementation by local agents. Care was taken to cover all vegetation types, but the network did not allow for a proportional coverage. However, we consider that this sampling design is a good compromise between methodological and implementation constraints, and should allow for reliable estimates of wildlife population abundance at the scale of the study site.

Sample survey

The survey was conducted from September to December 1999, during the hottest months (mean daily maximum temperature of 37 °C) preceding the start of the major rains (early January 2000). All counts were conducted in the morning. Each transect was covered once or twice a week (not on consecutive days), every week during the 4 months. We obtained an average of 30 replicates for each transect (min. 17, max.

47), spread homogeneously over the 4 months (40–60% of replicates occurred during September and October and 40–60% in November and December, according to transects). Animals were counted along transects by one or two observers on a bicycle at slow speed (6 km/h on average). The monitoring was focused on medium to large size mammals. All animals observed were identified and counted, and classified according to species, sex and age (classes = juvenile/adult). Observers assessed the distance of a detected animal (from the observer) by step counting from the centre of the line-transect to the position where the animal was first sighted. Each observer assessed conversion from steps to meters by stretching and adapting his steps to a meter. For animals that were potentially dangerous to approach on foot, such as large carnivores, elephant (*Loxondota africana*) or buffalo (*Syncerus caffer*), and when they did not flee after detection, the distance was estimated by the observer. As most of the species encountered in the area live in social herds, groups of animals were considered as the objects of measurement. Animals aggregated were counted in cluster. The centre of the group was approximately determined by each observer and used as the detection point for distance measurement.

Observers

All the observations presented in this survey were collected by 33 local agents after specific training for protocols. A locally trained co-ordinator in each Ward was responsible for following the survey in the field, with regular scientific and technical support from the Biodiversity Project team. Each observer was generally in charge of a single transect, but sometimes up to three. These observers were local agents from Anti Poaching Units, the Natural Resources Monitors from the Rural District Council and the local technicians called 'Barefoot Biologists', trained by the Biodiversity Project. They had all previously conducted patrols in the study area and had good knowledge of local wildlife.

Data analysis

Statistical analysis

Since distance of detection, encounter rate, and group size did not show a normal distribution, and inter-specific analysis dealt with a small sample size ($n = 10$ species), statistical analyses were run with non-parametric tests (SPSS 1997). Kruskal–Wallis (H) and Wilcoxon–Mann–Whitney (U) tests were used for comparison of means, and the Kendall test (r_K) was employed for rank correlations.

Line-transect analysis

Data analysis was carried out using DISTANCE software (Buckland et al. 1993). Clusters were the unit of analysis, and densities of individuals were calculated by the product of density of clusters multiplied by the average cluster size. To take into account the spatial and temporal heterogeneity in the data collected over various months and habitats, we used a stratification method. However, below we detail separately the different components of the estimation equation using the stratification approach.

First, a reliable estimation of the detection function requires a large sample size. Since the number of sightings for each species was low for each month or each transect, we pooled data across months and transects to estimate a global detection function, assuming detection function and effective strip width to be constant across strata. Analyses were run for each species where the number of sightings was above the recommended sample size ($n > 60$; Buckland et al. 1993), and for sable (*Hippotragus niger*) and bushpig (*Potamochoerus porcus*), despite their low sample sizes ($n > 40$), because of their value in terms of wildlife use, for trophy hunting (sable) and bush meat or crop raiding control (bushpig). We tested different choices of truncation distance values (if any) as well as data grouping to improve the model fit and probability of detection. Four standard models for the detection function were tested to fit data: uniform/cosine, uniform/simple polynomial, half normal/hermite polynomial and hazard rate/cosine (Buckland et al. 1993). Model selection was based on Akaike's Information Criterion and the χ^2 goodness of fit test.

Second, we carried out an analysis of the temporal (i.e. per month) and spatial (i.e. per transect) variability of the two other parameters of the estimation equation (i.e. cluster size and encounter rate) in order to select a stratification. Cluster size showed no significant variation across months, but showed a significant variation across transects for five species (buffalo $H_8 = 19.73$, $P = 0.011$; duiker *Sylvicapra grimmia* $H_9 = 41.49$, $P = 0.000$; elephant $H_8 = 20.43$, $P = 0.009$; grysbok *Raphicerus sharpei* $H_7 = 17.81$, $P = 0.013$; impala *Aepyceros melampus* $H_9 = 26.14$, $P = 0.002$). Similarly, encounter rate was constant across months for most of the species [except for bushpig, where encounter rate increased from September to December ($H_3 = 8.52$, $P = 0.036$), and for duiker, where it decreased between the same months ($H_3 = 11.10$, $P = 0.011$)]. Encounter rate was, however, highly heterogeneous across transects for all species (baboon *Papio cynocephalus* $H_9 = 44.04$, $P = 0.000$; buffalo $H_9 = 105.68$, $P = 0.000$; bushpig $H_9 = 27.46$, $P = 0.001$; duiker $H_9 = 25.68$, $P = 0.002$; elephant $H_9 = 48.87$, $P = 0.000$; grysbok $H_9 = 63.32$, $P = 0.000$; impala $H_9 = 31.00$, $P = 0.000$; kudu *Tragelaphus strepsiceros* $H_9 = 59.38$, $P = 0.000$; sable $H_9 = 51.29$, $P = 0.000$; warthog *Phacochoerus aethiopicus* $H_9 = 40.44$, $P = 0.000$). We therefore selected a stratification based on geographic stratum (i.e. transects). Following Buckland et al. (1993), we obtained a density estimate for each stratum by using a global detection function and by estimating cluster size and encounter rate individually by stratum.

Most of the species counted in this survey are herd-living species. It is often assumed that group size may influence the probability of sighting, with larger herds tending to be detectable at greater distances compared to smaller ones, thus causing an overestimation of the true density (Drummer and McDonald 1987). Consequently, for the analysis, we used a correction of the group-size bias for species whose observed cluster size was dependent on the detection distance. A mean expected cluster size was estimated with a size-bias regression method performed by DISTANCE when regression was significant at an α level of 0.15 [regression method = ln(cluster size)/estimated detection function $g(x)$], and the mean of observed clusters was used when it was not significant.

A global density was estimated for each species as the mean of all stratum

Table 2. List of mammals observed during the count, with numbers of sightings and individuals.

Species	Scientific name	Sightings	Individuals
Baboon	*Papio cynocephalus*	126	2080
Buffalo	*Syncerus caffer*	115	1502
Bushpig	*Potamochoerus porcus*	43	172
Duiker	*Sylvicapra grimmia*	186	260
Elephant	*Loxondota africana*	124	591
Grysbok	*Raphicerus sharpei*	109	144
Impala	*Aepyceros melampus*	201	1689
Kudu	*Tragelaphus strepsiceros*	171	654
Sable	*Hippotragus niger*	53	186
Warthog	*Phacochoerus aethiopicus*	74	329
African Wildcat	*Felis libyca*	24	30
Large-spotted Genet	*Genetta tigrina*	21	22
Eland	*Taurotragus oryx*	11	45
Burchell's Zebra	*Equus burchelli*	11	36
Bushbuck	*Tragelaphus scriptus*	9	15
Side-striped Jackal	*Canis adustus*	9	10
Vervet	*Cercopithecus aethiops*	8	71
Waterbuck	*Kobus ellipsiprymnus*	8	49
Lion	*Panthera leo*	7	20
Spotted Hyaena	*Crocuta crocuta*	5	6
Civet	*Civettictis civetta*	4	4
Antbear	*Orycteropus afer*	3	3
Wild dog	*Lycaon pictus*	2	14
Klipspringer	*Oreotragus oreotragus*	1	2
Leopard	*Panthera pardus*	1	1
Lesser BushBaby	*Galago senegalensis*	1	1
Porcupine	*Hystrix africaeaustralis*	1	2
Total	27 species	1328	7938

The 10 species listed in the upper part (Baboon–Warthog) were selected for density estimates.

estimates (i.e. transects) weighted by their stratum area. In order to measure each stratum area, we began by determining representatives of the major vegetation types crossed by our transects. We used a vegetation cover map (1:100 000 scale) produced by the Biodiversity Project from floristic surveys and a digital processing of SPOT images (De Wispelaere 2001) as a reference for vegetation type identification. First, the area surveyed along each transect was estimated as the transect length $\times$ the effective strip length estimated by DISTANCE $\times$ 2. Using GIS data analysis (Map Info Professional 6.5) we then determined the dominant vegetation types that covered each transect area. On each transect, we selected one or several dominant vegetation types that covered at least 85% of the transect area (two to four dominant vegetation types were selected per transect; Table 1). We also measured the total area covered by these selected vegetation types in the whole study area. We finally determined the stratum area as the sum of areas of the dominant vegetation types (i.e. their total area in the whole study area) we had selected in each transect (Table 1).

Table 3. Mean estimated densities D (ind./km^2) for the 10 most common species with their respective coefficients of variation (% cv).

Species	Model	Delta AIC	W	GOFχ-p	P	ESW	D	CV(%)
Baboon	U C	−0.02	100	0.99	0.63	63	6.55	15.6
Buffalo	U C	−1.10	120	0.79	0.60	72	4.18	22.4
Bushpig	U C	0.00	–	1.00	0.56	68	0.36	29.6
Duiker	Hn Hp	−0.41	100	0.36	0.52	52	0.78	10.5
Elephant	Hn Hp	−1.57	–	0.55	0.45	90	0.85	13.9
Grysbok	U C	−0.08	120	0.73	0.56	67	0.30	12.9
Impala	Hn Hp	−0.46	100	0.63	0.73	73	3.40	15.0
Kudu	Hr C	−1.97	–	0.74	0.51	102	0.96	12.4
Sable	Hn Hp	−1.90	–	0.76	0.80	104	0.23	27.6
Warthog	Hn Hp	−0.45	120	0.91	0.47	56	0.86	18.3

Detection function models (Hn Hp = Half-normal Hermite polynomial, U C = Uniform Cosine, Hr C = Hazard-rate Cosine) were selected among alternate models using AIC criteria. The value of the truncation distance W (in m) is presented for each species for which we deleted the outliers distance data. The delta AIC of models are presented (AIC value from selected model – minimum AIC from alternate model) as well as the probability of χ^2 goodness-of-fit test (GOFχ-p), the probability of detection (P) and the effective strip width (ESW) estimated (in m).

Population structure analysis

Clusters (i.e. groups) observed were considered to be representative of naturally occurring herds. The composition of these herds was examined in terms of group size, sex ratio (female adult/male+female adult) and age ratio (juvenile/female adult). A global group size was calculated for each species as the mean of cluster sizes estimated at stratum level (corrected or not with a size-bias regression method; see the line-transect analysis), weighted by the stratum area.

Results

Diversity

A total of 1328 observations were obtained, with 7938 animals counted (Table 2). The number of observations for all species combined was high in all transects (>100) except for one (Table 1). Observers identified 27 different species, mostly ungulates, which represented half of the species recorded and 75% of the total number of sightings. Several carnivore species were also recorded, but each with few sightings.

Line-transect density estimates

Different models were selected for density estimates for each species (Table 3). The sample size was large enough for the models to fit our data well. The effective strip width varied between species from about 50–100 m. It was not correlated with the mean observed cluster size ($r_K = -0.111$, $P = 0.655$, $n = 10$). However, it

Table 4. Population structure for the 10 most common species: the average group size (ind./group) is presented as the mean of the expected cluster size (ECS, cluster size corrected with the size bias regression method).

Species	Group size					Sex ratio	Age ratio
	ECS	SE	OCS	SE	Range (*n*)		
Baboon	14.68	0.03	14.94	0.03	40–1 (126)	0.66	0.36
Buffalo	7.66	0.08	7.94	0.08	79–1 (115)	0.65	0.54
Bush pig	3.63	0.02	3.89	0.02	14–1 (43)	0.63	0.35
Duiker	1.38	0.00	1.41	0.00	3–1 (186)	0.52	0.06
Elephant	3.48	0.02	4.04	0.02	17–1 (124)	0.69	0.50
Grysbok	1.17	0.00	1.26	0.00	3–1 (109)	0.51	0.17
Impala	5.96	0.02	8.36	0.03	32–1 (201)	0.64	0.30
Kudu	3.52	0.01	3.52	0.01	12–1 (171)	0.66	0.09
Sable	2.90	0.02	3.25	0.02	7–1 (53)	0.60	0.15
Warthog	4.48	0.02	4.48	0.02	9–1 (74)	0.65	0.43

The mean of observed cluster size (OCS) before correction is given for indication, as well as the range of the observed cluster size and the number of clusters considered (*n*). The composition of the population is described with the mean sex ratio (female adult/male+female adult) and the mean age ratio (juvenile/female adult).

increased significantly with species mean shoulder height calculated from Skinner and Smithers (1983) and Estes (1991) ($r_K = 0.556$, $P = 0.025$, $n = 10$). This may reflect a difference in visibility between species according to body size (Norton-Griffiths 1978).

The mean densities of individuals estimated by the line-transect analysis and coefficients of variation are shown in Table 3. Densities range respectively from 0.23 to 6.55 ind./km^2. Higher densities are found in baboon, buffalo and impala, but most of the species show a density lower than 1 ind./km^2. Coefficients of variation of density estimates are low (10.5 to 22.4%), except for the two species on which we gathered a lower number of sightings (bushpig and sable).

Population structure

The details of structure of populations are presented in Table 4. For most of the species the cluster size regression was significant. The mean expected cluster size varied consistently with the social behaviour of species, from solitary species (e.g. duiker = 1.38 ind./group; grysbok = 1.17 ind./group) to social ones (e.g. baboon = 14.68 ind./group; buffalo 7.66 ind./group). Age ratio (juvenile/female adult) varied from 6% (duiker) to 54% (buffalo). Sex ratio was even in monogamous species (duiker and grysbok), while it was skewed toward females in the polygamous species.

Observer bias

Since the monitoring involved a large number of agents, we tested the variation among observers in their ability to detect animals and to collect information. We made the comparison with all the data recorded by the different agents on the same transect. For each species, no significant difference was found on the size of cluster recorded by different observers. However, on 4 out of 10 transects there was a significant difference between the number of sightings collected by each observer (H_2 = 15.92, P = 0.000; H_3 = 18.03, P = 0.000; U = 5.50, P = 0.001; H_5 = 12.88, P = 0.025). One observer in particular gathered fewer observations than the others: he recorded 49% less sightings on average for the four transects monitored compared to those recorded by his colleagues. Moreover, on 4 of 10 transects, there was a significant difference between observers in the distance of detection reported (H_2 = 11.33, P = 0.003; H_2 = 30.02, P = 0.000; H_2 = 19.18, P = 0.000; H_3 = 13.08, P = 0.003). The difference was high, with a range from 10 to 52 m between observers according to transects. However, for the observers that had monitored several transects the bias was never the same, with their mean distance either above or below the average of the transect, hence included in the background noise of the data.

Discussion

Validation of the monitoring method

With a high number of observations collected during this first survey, bicycle counts appear to be an efficient technique for counting large mammals in a non-protected area. This technique has several advantages compared to other common wildlife census methods. First, it allows for an important census intensity. Bicycle counts are low cost compared to car or aerial counts, hence can be easily replicated and bicycles are faster than walking counts, therefore providing a higher ground coverage per human-hour labour, which contributes to maximise the sample size. Second, unlike car counts, bicycles ensure a silent approach, and are therefore especially appropriate in non-protected areas where animals are disturbed by the sight or the sound of a motor vehicle. In addition, observers on bicycles have a shape similar to a middle-size quadruped, which may reduce the fear generally induced by a human silhouette walking. Efficiency in animal detection is considered a real advantage for wildlife monitoring in communal lands where the encounter rate is often far lower than in protected areas.

Such advantages are illustrated by the list of species that have been observed during the survey: 27 of the 41 known resident species of the study area were recorded (Biodiversity Project 2002), including some of the most elusive (aardvark, lion and leopard) or rare species (wild dog). The non-observed species were mostly nocturnal or aquatic species (hippopotamus *Hippopotamus amphibus*, pangolin *Manis temminckii*, thick tailed bush-baby *Galago crassicaudatus* and small carnivores). This method proved to be especially appropriate for wild ungulates populations. All resident ungulate species were observed, apart from roan *Hippotragus equinus* (where only few herds are known to occur in this area), including some species locally uncommon, such as bushbuck, eland, klipspringer, waterbuck and zebra. In addition to abundance estimates of the populations of the most common species, the bicycle counts appeared to be a good technique to census species diversity.

Even in the thickest bushed areas, four-wheel-drive roads were straight and wide enough for an observer riding a bicycle to ensure that all animals on the transect line were detected. The roads were good enough for easy riding and allowed the observer to concentrate on the detection of animals. Bicycle movements are silent and relatively fast, hence animals can be detected at their initial location. Step counts for distance measurements are considered to be more accurate than visual estimation. They ensure reliable estimates of distance even if there may be a bias in step length and metric conversion between observers. In the case where observers had to estimate distance by eye for dangerous animals, experience acquired from foot measurements should reduce errors. We therefore considered the bicycle count survey to have met the main assumptions of a line-transect method.

The results we obtained on population structures were consistent with the ecology and social organisation of the species we observed (Estes 1991). Territorial and monogamous species, such as duiker and grysbok, showed an even sex ratio (respectively 0.52 and 0.51 F./Ad.) and a predominant individual social unit (observed group size = 1.41 and 1.26 ind./group, respectively). Gregarious species detected during the survey had a mean observed group size (e.g. impala = 8.40 ind./group, kudu = 3.82 ind./group) where values were consistent with other southern Africa sites (impala = 9.8 ind./group in November in Zimbabwe; Murray 1980; kudu = 3.5 to 5.1 ind./group; review by Perrin and Allen-Rowlandson 1992). Values of sex ratio fall within the range measured in other southern Africa areas (Jarman 1972; Ndhlovu and Balakrishnan 1991; Bothma 1996). However, values of age ratio obtained for some species (buffalo, duiker, elephant and kudu) seem to be erroneous, a result that might be explained by the lack of experience of local agents to distinguish age classes, for which such recording is unconventional. Apart from this point, the bicycle counts nevertheless appear to provide reliable information on the structure of wildlife populations, which is a useful complementary indicator of population status commonly used in wildlife management (Samuel et al. 1992).

This monitoring method benefits from the particular road network we opportunistically used, and may not be implemented with such ease in all areas. However, our results illustrate the fact that some simple ground-based methods can be very effective, and stress the need to develop locally adapted methodologies that take advantage of site-specific conditions (Caro 1999). Application of a rigorous ground-based method over vast areas is problematic, and aerial counts remain the only effective method for broad-scale surveys (Southwell 1996; Verlinden 1998). But the fine scale of resolution from bicycle counts is well suited to the purpose of local wildlife management. Among ground counting methods, indirect methods based on track or pellet counts have been widely used to provide indices of wildlife abundance. When animal detection is problematic, these methods may offer some advantages over methods that require sightings. They have been successfully attempted in African tropical forests where visibility is reduced (Koster and Hart 1988; Jachmann 1991; Walsh and White 1999; Plumptre 2000). However, relating sign numbers to animal abundance requires conversion factors to be calculated, and each has an associated error that affects abundance estimates (Plumptre 2000). Conversion factors associated with pellet counts (i.e. defaecation and decay rates) and their variability require a considerable effort to be measured upon numerous species (Koster and Hart 1988), and following Klinger et al. (1992) we consider this method unsuited to an area where observational surveys can be conducted. Despite the inherent variability of indirect estimation associated with track surveys, it could be a useful complementary method to monitor populations of secretive or low density species, such as large carnivores (Bertram 1979), for which it has proved to provide some reliable results (Beier and Cunningham 1996; Mills 1997).

As efforts were made to ensure simplification of the protocol and participation of numerous local observers, the counting method we implemented did present some limits on rigour of the sampling design (sampling area representation, number of replicates), as well as differences between observers' ability. These biases are considered difficult to measure. In this survey, we decided to pool the data collected from all observers in order to obtain a larger sample size. A high level of precision in density estimates was hence achieved (10–30% cv), which is always a key point in analysis in order to guarantee the repeatability of estimates (Hulme and Taylor 2000). A reliable inter-year comparison, however, requires these biases to either be constant over the year or to be corrected. Survey design should be standardised in terms of counting intensity, with a fixed number of replicates per month on each transect. The same transects should be used for successive surveys in order to improve the percentage of change in the population size detected (Plumptre 2000). At the same time, training of observers should be improved to try to standardise their ability of animal detection and measurements.

A high level of precision should remain the goal of implementation of a counting technique, but managers have to consider that only large changes will be detected. Special attention should be paid to these limits when wildlife management decisions are taken. The adaptation of harvesting plans (i.e. quota setting) to fluctuations of density estimates should be undertaken with caution. We recommend that quota schemes should be modified only after several years, to be effectively adapted to population abundance, in order to guarantee a sustainable exploitation of wildlife compatible with its conservation.

A participatory approach to wildlife management

Though community based conservation has been used throughout the world as the most appropriate approach to conserve wildlife, the success of its implementation has been questioned (Gibson and Marks 1995; Hackel 1999). A meaningful participation of rural residents in wildlife management with effective power in the decision-making process has been raised as a crucial issue to achieve successful conservation.

Counting large mammals with bicycles is an innovative monitoring method that meets the requirements for implementation based on active local participation. The field procedures are simple enough to ensure autonomous monitoring by local agents. Since it relies more on field experience than on technical or subjective concepts, it could be accessible to any bushwise community member. Moreover, implementation of the method relies on low costs, and is thus affordable to communities. Bicycles and their maintenance represent low investments. In this case study, most of the 33 observers that were involved were paid under CAMPFIRE funds for their common duties (anti-poaching patrols, natural resources surveys). Therefore, the technical and the financial autonomy offered by this monitoring method are crucial for a successful implementation in a long-term community based programme.

With the decentralisation of authority for rural resources through CAMPFIRE, local communities are already involved in the wildlife management decision-making process. The participatory approach developed in this study offers technical means to rural residents to obtain concrete data on wildlife resources through reliable and accessible information. It will reinforce the position of wildlife producer communities when debating options in resources management with safari operators and state institutional officers.

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