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Game and cattle trampling, and impacts of human dwellings on arthropods at a game park boundary

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Seven grassland sites were sampled at South Africa's Hluhluwe-Umfolozi Game Park boundary with the surrounding land, to assess changes in arthropod diversity in response to land use. Epigaeic arthropods were sampled using pitfall traps and a sweep net. In all, 262 morphospecies were collected, but this is an underestimate of total local species richness. Fifty percent of the species caught were single occurrences. The number of species, families and orders represented at each of the seven sites was not significantly different, but the number of individuals was significantly different. Between-site comparisons, using multivariate statistics, did not reveal any strong site groupings, with all sites being unique. The conclusion is that the reserve boundary does not significantly divide arthropod diversity on a simple inside-versus-outside basis. A major factor influencing the arthropod assemblages was intensity of land use. Indigenous game animals and domestic cattle had the same effect, and it was the intensity of trampling that was important rather than the type of trampling. Human settlements had a major impoverishing effect. The Coleoptera families, Cicindelidae, Staphylindae and Carabidae, were particularly sensitive indicator taxa of land use. Scarabaeidae species were the only group that were severely affected by the fence boundary, simply because their food source, the faeces of large native mammals, was inside, leaving them without resources outside the reserve.

Keywords: trampling; impacts; arthropods; Game Park; boundary.

Introduction

Lack of convergence between invertebrates and other organisms in nature reserves

Invertebrates are not necessarily protected when particular umbrella plants or vertebrates are conserved. Yen (1987), for example, found no significant relationship between the number of vertebrate and Coleoptera species in each vegetation type. This means that an area reserved on the basis of a rich or unique vertebrate fauna may not necessarily have the same value for Coleoptera. In general, there seems to be little correlation between invertebrates and other animal taxa (New, 1987; Prendergast, *et al.*, 1993). This suggests that invertebrates require research *per se*, with no assumptions that they are protected when megavertebrates are conserved.

Reserve boundaries

The fencing of game reserves leads to different land uses on the inside compared to the outside, as a result of high megaherbivore diversity and density inside and high human pressure on the outside. It would be interesting to know how this divisive land use affects the distribution of invertebrates, the major components of biodiversity. Invertebrates often have much smaller home ranges than the vertebrates and their beta diversity is relatively higher.

The reactions of terrestrial invertebrates to microenvironmental gradients (such as temperature and humidity, which are, in turn, affected by land use) make them highly responsive to edge effects (Unwin and Corbet, 1991). Rapid population growth rates and short generation times mean that invertebrates show rapid responses to environmental changes, making it possible to do short-term spatial studies of invertebrate diversity to detect these changes (Majer, 1987; Kirby, 1992; Kremen *et al.*, 1993; Samways, 1993).

Aims of this study

Here, we look at the quantitative changes in arthropod diversity in response to environmental changes at the boundary of the Hluhluwe-Umfolozi Game Park, an important world rhino reserve.

Classifying individuals into morphospecies or Recognizable Taxonomic Units (RTUs) is a mechanism for rapid biodiversity assessment (Beattie and Oliver, 1994). RTUs can be used in ecological impact assessments and eliminate much time normally spent on species identification (Oliver and Beattie, 1993). Using RTUs as a first step, and then taxonomic identifications where possible, the following questions were addressed.

(1) Is there a decrease in arthropod diversity and abundance outside Hluhluwe-Umfolozi Game Park through heavy human land use?
(2) Is there a decrease in arthropod diversity and abundance immediately inside the fence boundary where the confined large mammals showed increased trampling?
(3) Does the reserve boundary act as a filter or block for arthropods as it does for the large mammals?
(4) Are there sensitive indicator groups to landscape changes in this savanna ecosystem?
(5) Can the sites be grouped on a binary inside/outside basis according to arthropod species composition, or is each locale unique in its own right irrespective of the reserve boundary?

Materials and methods

Study area

The study area was at the southern boundary of Hluhluwe-Umfolozi Game Park, in KwaZulu-Natal, South Africa: 28°15′S 32°00′E to 28°10′S 32°06′E.

Site location and description

The seven sites were grassy patches in a matrix of open *Acacia* savanna. The nomenclature, position and land use for the sites are given in Table 1.

Table 1. Summary environmental data for the seven sites

Site[a]	Distance from boundary (m)	Grass height (cm)	Inflorescence height (cm)	Dominant grass	Use
1	150	20–25	60–70	*Sporobolus africanus* (Poir.)	Domestic dwellings
2	150	20–30	50	*Melanus repens* (Willd.)	Cattle
3	6000	10	40	*Digitaria* sp.	Cattle/crops
4	200	15	55	–	Game
5	1200	15	60	–	Game
6	5500	10–15	80	*Panicum dregeanum* (Nees)	Game
7	4000	40–45	60	*Setaria* sp.	Game

[a] Site 1 = outside the reserve, at the reserve boundary; Site 2 = outside the reserve at the reserve boundary; Site 3 = outside the reserve, far from the boundary; Site 4 = inside the reserve, at the reserve boundary; Site 5 = inside the reserve, at the reserve boundary; Site 6 = inside the reserve, far from the boundary; Site 7 = inside the reserve, far from the boundary.

Sampling methods

Sampling was by two well-established methods (pitfall trapping and sweep-netting) with their well-known advantages and disadvantages (Greenslade, 1973; Southwood, 1978; Samways, 1990). At each of the seven sites, six small, and three large pitfall traps were sunk. The small traps were 18 × 150 mm glass test tubes in PVC sleeves (Majer, 1978) and the large ones were plastic honey jars with a lip diameter of 50 mm. The traps were at 2 m intervals. The small traps were sunk linearly next to each other to form one line, going from left to right across the site. The large traps were sunk perpendicular to the small traps so that the traps formed a ‘T’.

A 1 : 3 mixture of glycerol: 70% alcohol was used as the fixative in the traps. It is not attractive, at least to ants (Greenslade and Greenslade, 1971).

Sampling was on consecutive three-day intervals, but no longer, to avoid the traps overfilling with rainwater or specimens. The study period was an intensive three-week interval over January 1994. Each site was swept 100 times with a net (φ = 325 mm) for 10 min per site at the upper levels of the grass.

The following environmental variables were measured at each site: vegetation height, aspect and slope, soil temperature, soil relative humidity, percentage bare ground, and percentage forb cover. These data were used in subsequent multivariate analyses.

Soil temperature and soil relative humidity were measured six times per site using a hygro-thermometer (Thies, Clima, Germany no. 1.0466.00.000). Aspect was determined using a compass, and slope was estimated using a protractor. Percentage bare ground and percentage forb cover were estimated using 10 × 1 m^2 quadrats per site.

Grasses were identified using van Oudtshoorn (1992), and a presence/absence species list was compiled for each site.

Analysis of data

A raw data matrix showed arthropod species diversity and relative abundances for each site. All insects were classified to family using the keys in Scholtz and Holm (1989). The data from each sampling date were used to give cumulative figures. These data are presented quantitatively using histograms. Chi-square tests (one-way contingency tables) were performed to test for differences between sites for number of individuals, species, families and orders. Expected values were calculated as mean values for each site.

Qualitative differences were analysed using the Fortran programmes DECORANA (Hill, 1979a) for ordination and TWINSPAN (Hill, 1979b) for cluster analysis. One can have the best of two perspectives (classification and ordination), by presenting a two dimensional ordination of the data under investigation and then drawing partitions that yield the classification directly on the ordination scatter diagram (Pielou, 1984). Sites were also compared using Hill's (1973) diversity numbers and evenness measures (Ludwig and Reynolds, 1988). Hill's N_0 is the total number of species at a site, while N_1 is the number of abundant species, and N_2 is the number of very abundant species. The closer N_2 is to one, the more the species assemblage is dominated by a single species (Ludwig and Reynolds, 1988). To determine the effect of environmental variables (soil temperature, relative humidity, percentage bare ground and percentage forb cover) a Principal Components Analysis (PCA), ANOVAs and Scheffe tests were performed, all using STATGRAPHICS (version 5.0). Data that were presented as percentages were arcsin-transformed before analysis (Alder and Roessler, 1968).

Results

A total of 262 morphospecies and 6996 individuals were caught at all sites combined. Quantitatively, the sites did not differ greatly from each other in number of species, families and orders represented at each site. Chi-square tests ($p < 0.05$; 6 df) showed no significant differences between sites as measured by species richness, families or orders, but the sites did differ significantly in numbers of individuals caught ($p < 0.05$; 6 df). Species from the main orders were graphed (Fig. 1) and again the sites appear to be very similar.

There were many rare species and few abundant ones, with 49% of the species having single occurrences. Comparing the number of single-site species (considered here as stenotopic) revealed no significant difference between sites ($p < 0.05$; 2 df).

Species overlaps for each site gave an indication of stenotopism versus eurytopism (Fig. 2). A linear relationship (log average number of individuals per site) = $-0.99 + 0.30^*$ site overlap; $p < 0.05$; $r = 0.813$) existed between the mean number of individuals per site and the number of sites at which these species occurred. Most species were stenotopic with low abundances, with only a few being eurytopic with high abundances.

Graphing the species that occurred at single sites under their respective orders (Fig. 3) showed that Coleoptera were the most stenotopic, and Orthoptera the most eurytopic (of the major orders).

Hill's diversity numbers and evenness values showed that site one was dominated most by a single species, and site six the least (Table 2). Evenness values were similar, ranging between 0.309 and 0.385, except site 1, which had a value of 0.539 (Table 2).

For the grasses, the TWINSPAN analysis was complete at level two (Fig. 4a), but for the

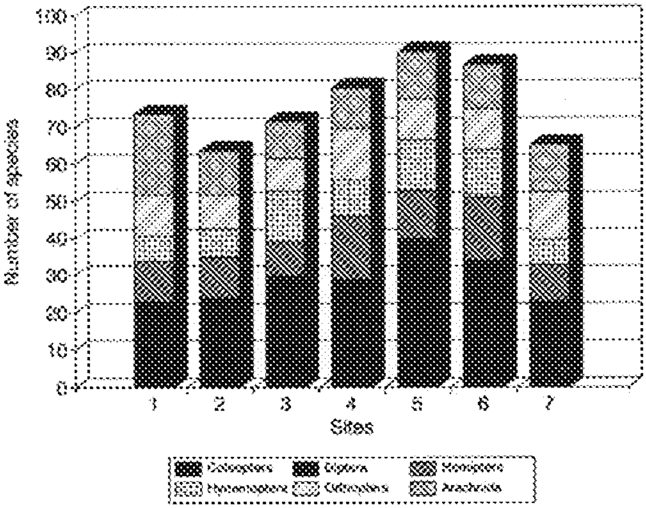

Figure 1. Species richness of the main taxa at the seven sites.

insects, it was complete at level three (Fig. 4b). Thus, the classification derived from the arthropods was slightly finer than that for the grasses. The ordination using DECORANA (Fig. 5) gave a similar grouping of the seven sites, showing distance relationships between sites.

To explain the DECORANA groupings, sites were analysed against environmental variables (forb cover, bare ground, temperature and relative humidity) using Principle Components Analysis (Table 3, Fig. 6). The two most important components were bare

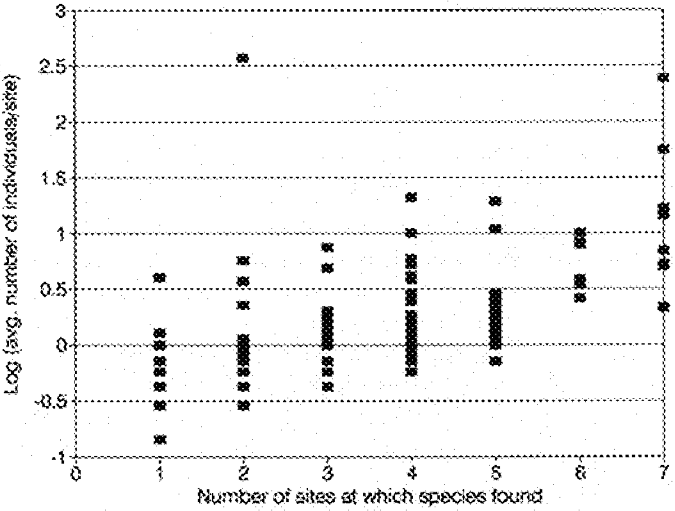

Figure 2. Stenotopism versus eurytopism: species overlap at the seven sites. On the x-axis, the numbers represent species occurring at one site, two sites, and so on up to species common to all seven sites.

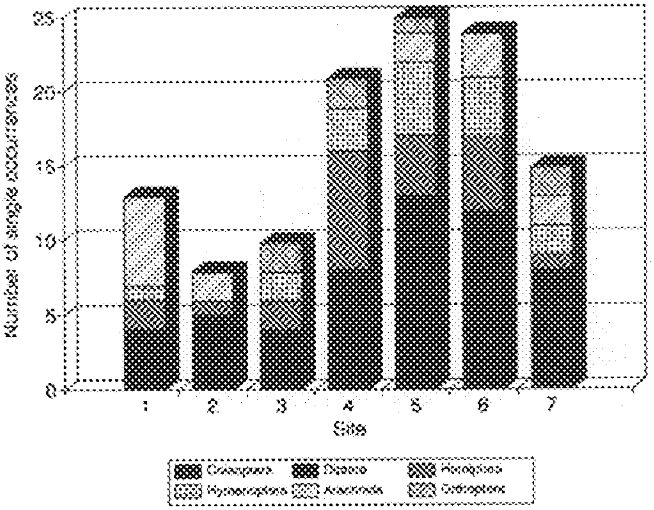

Figure 3. Numbers of species (among the six main taxa) that occurred only at any one of the seven sites. Notice that the Coleoptera contributed many species that were specific to one of the seven sites, while the Orthoptera contributed the fewest.

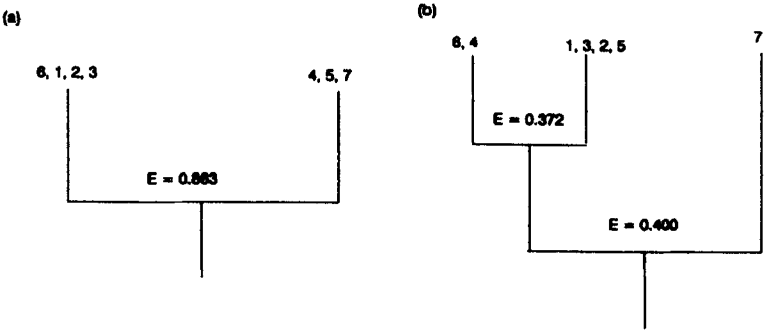

Figure 4. TWINSPAN classification of the seven sites on the basis of, (a) presence/absence of grass species, (b) arthropod species composition. E = eigenvalues.

Table 2. Hill's diversity numbers and evenness measures

Site	N_0	N_1	N_2	E^a
Site 1	86	2.549	1.375	0.539
Site 2	77	11.056	4.138	0.374
Site 3	79	13.082	5.042	0.385
Site 4	90	10.728	3.525	0.329
Site 5	103	14.229	4.393	0.309
Site 6	97	30.088	10.843	0.360
Site 7	79	9.306	2.931	0.315

[a] $E = N_2/N_1$

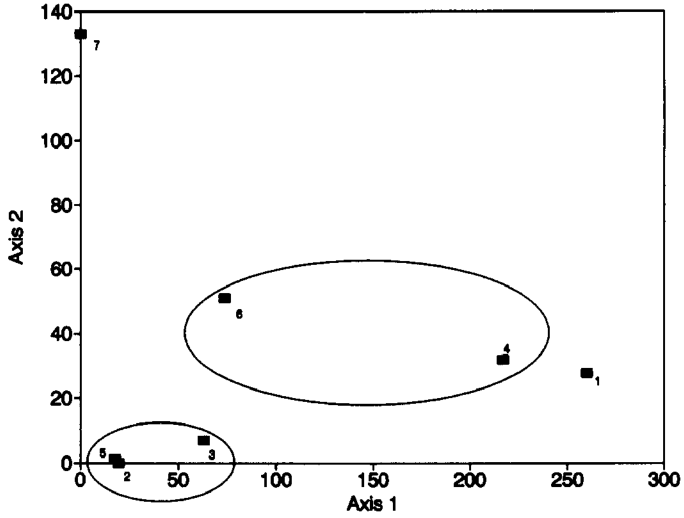

Figure 5. DCA ordination of the seven sites based on all species sampled. Note that these groupings were achieved by using the groups from the TWINSPAN classification in Fig. 4b.

ground (component 1) and forb abundance (component 2). To determine whether these variables were significant in explaining site positioning, one-way ANOVAs using Scheffe tests were performed. Between-site differences in soil temperature ($F_{6.35} = 0.784; p > 0.05$) and relative humidity ($F_{6.35} = 1.268$; $p > 0.05$) were not significant, but they were significantly different for forb abundances ($F_{6.63} = 16.024; p < 0.01$) and percentage bare ground ($F_{6.63} = 13.400; p < 0.01$). However, multiple range analysis did not separate out sites, and there was not any explainable pattern.

The Coleoptera families, Scarabaeidae, Curculionidae, Cicindelidae, Carabidae and

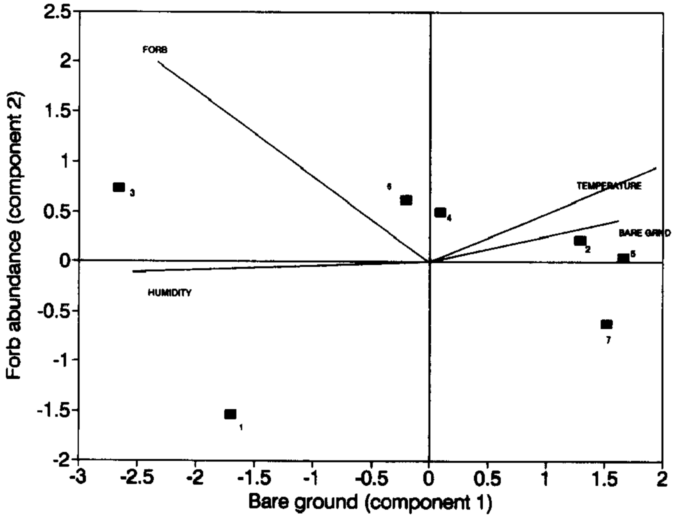

Figure 6. Principal components biplot for main environmental variables.

Table 3. Weightings of environmental variables on the first two axes of a Principal Components Analysis

Variables	PC Axis 1	PC Axis 2
Eigenvalue	2.78	0.66
Percentage variance	69.61	16.46
Cumulative percentage variance	69.61	86.07
Bare ground	0.46	0.19
Forb cover	−0.42	0.88
Temperature	0.53	0.42
Relative humidity	−0.58	−0.10

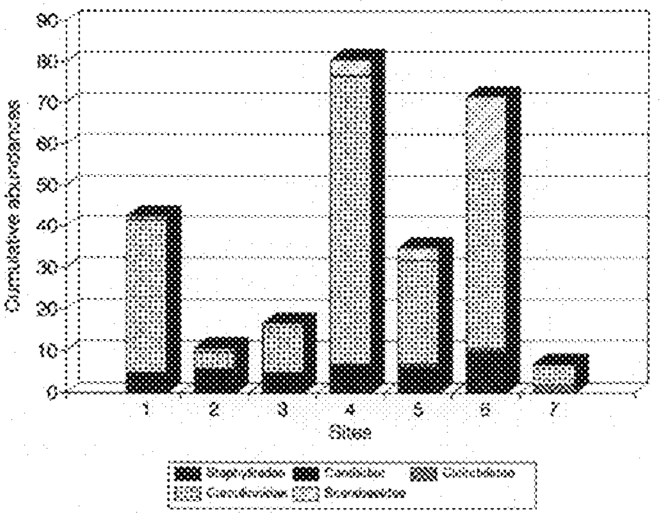

Figure 7. Abundances of the main Coleoptera families at the seven sites.

Staphylinidae, are potentially sensitive indicator taxa (Fig. 7). However, no distinct trends emerged for the different sites. Of the five families chosen, site one had three; sites two, three and seven had two; sites four and six had four, and site five all five. Again, sites did not separate out clearly.

Discussion

Quantitative analysis

The sites were not significantly different in numbers of RTUs, making the sites equally species rich.

The high number of individuals at the grass site just outside the reserve (site one) is explained by the very high numbers of one species of ant (*Anoplolepis custodiens* (Smith)).

In the case of ants, landscape disturbance loses the rare species and promotes the common ones (Samways, 1989), as verified here.

A species-rich site is likely to contain many more localized and rare species than widespread and abundant ones (Stork, 1993). Indeed, the majority of insects in this study were rare. The number of single-occurrence species at each site is slightly higher at sites inside than outside the reserve, but this difference was not significant. At least in terms of species richness, arthropod diversity outside the reserve was not lower than inside.

Intensity versus type of land use

To determine total species richness at any one site may require long-term sampling, possibly over several years (Owen and Owen, 1990). However short-term studies using appropriate sampling methods are nevertheless useful for making local spatial comparisons.

Using Hill's numbers and evenness measures, it was apparent here that the 'grass site just outside the reserve' (site one) was the least diverse community. This may be explained by the intensive human land use and the presence of dwellings. All the other sites were either low-intensity agriculture (outside) or under game management (inside). This suggested that it was the intensity of land use which was the overriding factor. In other words, whether grazing was by game or by cattle did not matter, unless it became too intense, or the land was transformed by human dwellings.

Effect of trampling

In the PCA, percentage bare ground explained most of the site variation, and forb abundances the next most. The ordination of sites 'next to and outside the reserve fence' (site two) and 'next to and inside the reserve fence' (site five) were explained most by bare ground. Both these sites, being at the reserve boundary, were fairly degraded through trampling, albeit by cattle outside and game inside.

Sites 'grass inside reserve' (site six) and 'grass inside reserve next to track' (site four) grouped together possibly as a result of forb abundance. These two sites were trampled but not degraded. The other sites were all unique, which is not surprising as insect diversity is sufficiently high to make any individual patch ultimately unique (New, 1987).

Use of arthropods versus grasses to classify sites

The multivariate site classification using arthropods gave a much finer classification than that using grasses. Similarly, Yen (1987) found that vegetation classified using Coleoptera was finer than that using vertebrates. Coleoptera are more specific to particular vegetation communities, and are richer in species than are vertebrates.

Indicator species, conservation targets, and the importance of Coleoptera

The commonest arthropods are of less value for conservation evaluation, while the intermediate ones can be good indicators (New, 1987). Rare, stenotopic species, by virtue of their scarcity, are also of less value, but nevertheless may be important conservation subjects. However, with short-term sampling there is always the possibility that an insect is rare or absent through a temporary drop in its population level rather than from disturbance (Samways, 1994).

Since the sites did not differ significantly in the number of rare species (i.e. one individual at one site), from this point of view, all the sites were of equal conservation status. Furthermore, the Coleoptera was the major taxon in this study using the chosen sampling methods, and the one with the highest percentage of stenotopic species (Fig. 2), illustrating its large contribution to the biodiversity of the area, as in other areas (Stork, 1993).

Diversity *per se* is not sufficient on its own for choosing reserves, as it may be increased by numbers of generalists or tourists following disturbance. In a country like South Africa, with many local endemics, it is important therefore to recognize which species are making up the community (Ackery and Vane-Wright, 1984).

Specific indicator species can also be used to verify the multivariate statistics which infer the degree of degradation of the sites. Many arthropods are illustrative for this, as they respond rapidly to environmental changes through their sensitivity to microclimate, mobility to find the right microclimate, and their short life cycles. When different taxa all respond in the same way to these changes this can be taken as an indication of environmental change (Kremen *et al.*, 1993). For landscape evaluation, diversity is widely accepted as a good criterion, even if morphospecies or RTUs are used instead of named species (Beattie and Oliver, 1994). However, assessing total species richness for conservation purposes is difficult because of the logistics of sorting a vast number of often unidentifiable species, and also because it is never clear when the last species has been sampled. The best approach is to concentrate on a small number of insect groups which can act as indicators or umbrellas, but at the same time knowing the drawbacks of this approach (Majer, 1983; Andersen, 1987; Davis *et al.*, 1987; Pearson and Cassola, 1992; Samways, 1994).

This study highlighted the Coleoptera as a particularly overall sensitive taxon to landscape change. The choice of Coleoptera here also supports the findings of Davis *et al.* (1987), Pearson and Cassola (1992) and Yen (1987). Besides Coleoptera however, this study also illustrated that Diptera, Hemiptera, Hymenoptera, Orthoptera and Arachnida are also sensitive indicators.

Within the Coleoptera, the families Scarabaeidae, Staphylinidae, Cicindelidae and Carabidae form a potentially sensitive group of taxa which covers a wide trophic range. Furthermore, these species in these families are relatively easily identified, and have also been used by other workers, for example Carabidae: Turin and den Boer (1988), Niemela *et al.* (1993), Stork (1993); Scarabaeidae: Klein (1989); Cicindelidae: Pearson and Cassola (1992); Clark and Samways (1995).

From Fig. 7, using these families, there is no difference between inside and outside the reserve. Furthermore, the small differences relating to particular species may be the result of natural variation in their distributions rather than to environmental change. However, all species of Scarabaeidae, except for one are found exclusively inside the reserve. This is a response to the outside depletion of indigenous megaherbivores, with domestic cattle dung generally not being sufficiently attractive to them. The Curculionidae species were highly eurytopic and did not reveal anything about site differences or degree of environmental degradation.

Conclusions in response to the questions posed, and management recommendations

(1) There is no significant decrease in overall arthropod diversity and abundance outside the game park, except where there were human dwellings. This anthropogenic impact supports the concept of 'Multiple-use-Modules' (Noss and Harris, 1986) and 'Man-and-the-Biosphere' reserves with core areas without dwellings.

(2) Arthropod diversity decreased closely adjacent to the park boundary fence, because the fence directed the megaherbivores (wild game inside and cattle outside) along the edges.

(3) The boundary is not a filter or block for the arthropods as it is for the large mammals. The important exception is the Scarabaeidae. In all species but one, they were confined to inside the reserve simply because their right food was available there.

(4) Coleoptera, especially the families Staphylinidae, Carabidae, and Cicindellidae, are potentially good indicators of intensity and type of land use.

(5) Sites cannot be grouped simply on a binary inside/outside basis, but on particular type and intensity of land use, e.g. dwellings *versus* non-dwellings, low-intensity trampling (whether by game or cattle) *versus* high-intensity trampling.

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