CHAPTER 3

Using broad-scale environmental surrogate measures for selecting viable populations of "umbrella species" and regional biodiversity

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1. INTRODUCTION

Nature conservation has only recently become recognised as a legitimate form of land use (Margules and Usher, 1981), competing for limited land resources with agriculture and forestry, as well as urban and industrial development. Conservation areas play an indispensable and vital role in the protection of natural ecosystems (regional biodiversity). However, more biodiversity exists in agricultural, pastoral, forestry and other human-dominated ecosystems, that cover about 80 percent of the world's terrestrial environment, than in protected areas (World Resources Institute, 1993). A central issue in conservation biology is the importance of conserving maximum biological diversity in the limited land surfaces available to nature conservation (Freitag *et al.,* 1996). Critical decisions regarding the permanent location of protected areas are presently being made, thus emphasising the need for reserve networks to be as representative of biodiversity as possible, i.e. containing as many elements of biodiversity as possible (Pressey *et al.*, 1993).

Identifying areas with high conservation values, which are both representative of biodiversity and complements the current reserve network, requires extensive species distribution information (Pressey *et al.*, 1993). Since adequate databases on the distribution of species seldom exist (Nicholls, 1989; Belbin, 1993; Haila and Margules, 1996), the use of surrogate measures for biodiversity has to be relied on in systematic conservation area selection procedures. In the present study we aim to establish if viable populations of large herbivore species can be incorporated into conservation area selection procedures by using broad-scale surrogates and to ascertain how efficient these surrogates are at including finer scale elements of the biodiversity estate. The aims are (1) to evaluate the manner in which the selection of varying degrees of surrogate variables incorporates viable populations of the large herbivore species in the Kruger National Park (KNP) and (2) to determine whether vegetation type as a broad scale surrogate represents unsurveyed species by evaluating the species representation of this surrogate at a broader scale (Savanna biome) using existing presence/absence data on seven other taxa (mammals, birds, butterflies, termites,

antlions, buprestid beetles and scarab beetles). These specific taxa were used because they have previously been used in conservation area selection procedures where a species-based approach was used and the insect groups are representative of four insect orders with diverse habitat requirements.

The concept of biological diversity first appeared in the ecological literature towards the middle of the 1980s (Ghilarov, 1996), usually in the context of concerns over the loss of the natural environment and its contents (Gaston, 1996). Various definitions for biodiversity have been proffered, all broad ranging, imprecise and providing a poor foundation for its practical measurement (Gaston, 1994). The practicalities of conserving maximum biological diversity will greatly depend on whether or not we can find effective units of measurement for biodiversity itself for the purposes of conservation planning (Soulé, 1989; Vane-Wright *et al.* 1991; Pressey *et al.*, 1993).

A wide variety of different possible measures of biodiversity have been suggested, applied and discussed (e.g. Vane-Wright *et al.,* 1991; Gaston, 1994; Gaston,1996). One possible solution to this problem of biodiversity-measurement is to find appropriate indicators for these measures. Indicators are measurable surrogates for biodiversity or other environmental endpoints (Noss, 1990). Such surrogates should be quantities that are more easily determined and which correlate strongly with those measures of biodiversity which are ultimately targeted (Gaston and Blackburn, 1995). According to Gaston and Blackburn (1995), three groups of surrogates have been suggested in the past: (i) species richness of an indicator group, (ii) the numbers of higher taxa and (iii) the diversity of various broad environmental parameters.

Broad scale environmental parameters

Systematically selecting regional conservation areas in order to represent all surrogate classes - and consequently the region's entire range of environmental variation - assumes that these areas contain all the species found in that region (Belbin, 1993; Nicholls and Margules, 1993; Faith and Walker, 1996). Through a pattern-based approach such as this, species variation is linked to environmental variation as summarised in an environmental

pattern. Wessels *et al.* (1999) reviewed this rationale, and found that the environmental representativeness of reserve networks have been assessed using a variety of surrogate measures, including climatic attributes (Austin and Nix, 1978), climatic and edaphic variables (Belbin, 1993), landscapes (Pressey and Nicholls, 1989), landform-vegetation classes (Awimbo *et al.*, 1996), land systems (Pressey and Nicholls, 1989), landscape ecosystems (Lapin and Barnes, 1995), environmental groups (Mackey *et al.*, 1989), environments (Margules *et al.*, 1994) and environmental domains (Bedward *et al.*, 1992).

The highly controversial issue of population viability is one that will unlikely be resolved in the immediate future. There are two aspects associated with minimum viable population size: genetics and population dynamics. Two possible estimates for the size of populations that would be needed to conserve the genetic variance within the population is offered by Franklin (1980) - the much quoted 50/500 controversy. Animal production studies indicate that inbreeding is kept to a tolerable level if the population consists of 50 individuals. In natural populations, on the other hand, the lower limit of 500 individuals is suggested by Franklin (1980). Since this is an effective population size, the actual size of the population would be three to four times larger in order to score $N_e = 500$. Caughley (1994) argues that Franklin's 500 relates to the amount of genetic variance one might wish to retain - 500 indicating the amount of genetic variance that is equal to the amount of environmental variance expressed in the phenotype of a totally homozygous population. In a study on Drosophila it was shown that this variance would be retained by a genetically effective population of 500.

If a population conforms to the minimum conditions for the long-term persistence and adaptation in a given location, it can be termed a viable population. Long-term persistence can be defined in this context as the capacity of the group to maintain itself without significant demographic or genetic manipulation for the foreseeable ecological future (usually centuries) with a certain degree of certitude, say 95%. The probabilistic qualification is necessary, because it would be impossible to absolutely guarantee the survival of a group, no matter what the size. The long-term viability of a group does not depend solely on size and population dynamics - factors like environmental variation and

stochasticity, genetics, catastrophes as well as metapopulation structure and fragmentation play a vital role in the persistence of a group. A minimum viable population (MVP) can thus be thought of as a set of estimates that are the product of a systematic process (a population viability analysis) for estimating species-, location- and time-specific criteria for persistence and survival (Soulé, 1989). Thus, no single value or magic number exists that has universal validity. The subjective choice of an acceptable level of risk will determine the numbers, densities and distribution in space of a MVP, because every situation or population is unique.

McNab (1963 *loc. cit* Holling, 1992.) found that the home ranges of mammals scaled in the classic allometric relation to body mass. Similarly, Holling (1992) concludes that the spatial grain at which a landscape is sampled by animals is largely independent of both the landscape and the animal's trophic status, but it is a function of body size, and presumably, of body form. Thus, the larger the body size, the larger the home range of the animal. It can thus be accepted that as an umbrella component of regional biodiversity, any surrogate procedure which effectively selects viable populations of large herbivores, with associated large home ranges, will maximise the probability that most other biodiversity components are sufficiently included into a reserve network, since smaller species generally require smaller ranges. This notion will be tested at a local scale, namely for the Savanna biome, by using available data on the distribution of seven other taxa.

2. METHODS

a) Study area

The study area comprises the Kruger National Park (KNP) located in the Northern and Mpumalanga Provinces of South Africa, encompassing an area of roughly 20 000km^2. It is situated in the Savanna biome of South Africa, and consists of seven different Savanna vegetation types (Low and Rebelo, 1996). The mean annual rainfall in the Park, measured over a period of 73 years (1919/20 – 1992/93), is 534mm. Long-term mean temperatures for this area range between 15.8^{O}C and 29.7 ^{O}C over the same period of time (Zambatis and Biggs, 1995).

b) Animal abundance and distribution data

Point data for the period 1981-1992 obtained from the Kruger National Park annual ecological aerial census for 12 large herbivore populations were reclassified into $4km^2$, $12.5km^2$ as well as $25km^2$ grid square cell layers respectively. The twelve large, unmanaged herbivore species occurring in the KNP used in the study, are impala *(Aepyceros melampus)*, blue wildebeest *(Connochaetes taurinus)*, zebra (*Equus burchelli)*, white rhinoceros *(Ceratotherium simum)*, giraffe *(Giraffa camelopardalis)*, kudu *(Tragelaphus strepsiceros)*, sable antelope *(Hippotragus niger)*, eland *(Taurotragus oryx)*, warthog *(Phacochoerus aethiopicus)*, waterbuck *(Kobus ellipsiprymnus)*, tsessebe *(Damaliscus lunatus)* and the roan antelope *(Hippotragus equinus)*.

Data from four years were used to assess whether differences in habitat quality will influence the inclusion of viable populations. Two of the years had an above average rainfall, with $\bar{x}$ = 774mm/annum (1981 and 1985) and for two years (1983 and 1992) a below average rainfall figure was recorded ($\bar{x}$ = 267mm/annum).

c) Environmental surrogates

Within the KNP a variety of different scaled land classification systems have been developed, namely: land systems (Venter, 1990), land types (Venter, 1990) and landscapes (Gertenbach, 1983). These classifications, together with vegetation types (Low and Rebelo, 1996) were used as broad-scale surrogates in the present study. Environmental data can be used to assess the relative biodiversity of an area, because if the environmental variation (or habitat/ecosystem variation) is measured correctly, it should be a good indication of species diversity (Faith and Walker, 1996). These four different land classification systems were employed to explore the consequences of using environmental surrogate measures for conserving viable populations of large herbivores. Using ArcInfo® GIS, Version 7.1.2 (ESRI, Inc., Redlands, California, USA) each of these four land classification systems were intersected with each of the three grid cell layers respectively, resulting in 12 scale combinations used in the present study (Table 1). Through these intersections, the proportion of each unit in a classification system occupying all grid cells in a layer was obtained.

Table 1: A summary of the scale combinations used

Land type (Venter, 1990)	Land system (Venter, 1990)
Land type and 4km^2 grid	Land system and 4km^2 grid
Land type and 12.5km^2 grid	Land system and 12.5km^2 grid
Land type and 25km^2 grid	Land system and 25km^2 grid
Vegetation type (Low & Rebelo, 1996)	Landscape (Gertenbach, 1983)
Vegetation type and 4km^2 grid	Landscape and 4km^2 grid
Vegetation type and 12.5km^2 grid	Landscape and 12.5km^2 grid
Vegetation type and 25km^2 grid	Landscape and 25km^2 grid

The land system classification (Venter, 1990), comprising 11 land systems, was developed on the basis of geology, geomorphology and broad climatic attributes. These land systems were further classified according to soil type, vegetation type and landform into 56 land types (Venter, 1990). Thirty-five landscapes (Gertenbach, 1983) were identified according to specific geomorphology, climate, soil, vegetation pattern and associated fauna. Vegetation types (Low and Rebelo, 1996) are defined as those units that have a similar vegetation structure, sharing important plant species and having similar ecological processes.

d) Surrogate selection

An algorithm was applied to randomly select a fixed percentage of each of the land classification units of each surrogate, and the number of individuals of each species occurring in the selected areas was quantified. This was to see how, and if, viable populations of the large herbivore species are included in the selected areas. This analysis was performed on each of the four data sets (1981, 1983, 1985, 1992) and at each of the 12 scale combinations (Table 1). Thus a total of 48 different data sets were assessed using the Percentage Area Representation algorithm (Wessels *et al.*, 1999).

Although this algorithm (hereafter referred to as PAR) is land-use efficient, it provides an invariable result with the same outcome at each run, considering that the starting point is

always the grid cell containing the largest area of the smallest land unit. Therefore, an algorithm that will perform a specified number of iterations at each representation level (e.g. 10%, 20% ...50% of each vegetation type), in order to generate alternative networks of grid cells, was coded. The PAR algorithm was modified, resulting in an algorithm where multiple iterations are possible, and which requires a randomly arranged list of each land type and a pre-specified random initiation grid cell. This algorithm (PARI) was used only for the land type classification, to determine whether changing the starting point and subsequent selection rules, will have any influence on the results. Average percentages of individuals included through the land type selection were calculated. The selection rules of both these algorithms are provided in Appendices A and B respectively.

e) Habitat quality

Provided that differences exist between the results obtained for the four years in the previous analyses, it would be an indication that rainfall, and hence habitat quality, influences species abundance. To deduce whether changes in habitat quality indeed affect included population sizes and the consequent spatial distribution of individuals across the study area, we tested for significant differences between the data derived from the four years. As habitat quality and species density in one year influences following years, these data are not independent. Kendall's Coefficient of Concordance (Zar, 1996) was therefore used.

$$W = \frac{\sum R^2 i - [\sum (Ri)^2] / n}{[M^2 (n^3 - n)] / 12}$$

Correlation, or association, between more than two variables can be measured nonparametrically by Kendall's coefficient of concordance. Ranks for each of the variables have to determined from frequency distributions, and these distribution data ranked according to Kendall's method to obtain Ri values, where Ri is the sums of ranks, M is the number of variables being correlated and n is the number of data per variable. A W-value close to one indicates high concordance (association) between the different data sets, and the closer this value gets to 0, the less association exists between data sets.