# A multicriteria assessment of regional sustainability options in the Northern Province, South Africa.

by

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*"And as the natural man within loses honour,*

*so too does nature without.*

*We no longer feel reverence for nature, and defoliation of spirit and landscape are everywhere to be seen...*

*That is why what is left of the natural world matters more to life*

*now than it has ever done before.*

*It is the last temple on earth which is capable of restoring man to an objective self wherein his ego is transfigured and given life and meaning without end..."*

Laurens van der Post, *Feather Fall*. (1994).

**A multicriteria assessment of regional sustainability options in the Northern Province, South Africa.**

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**Abstract**

The need to protect biological diversity, the inadequacies of current protected areas and the need for scientific procedures for the identification of areas important to biodiversity conservation are well-known facts in conservation biology. Many conservation planning techniques developed, however, have a number of weaknesses. These shortcomings include incomplete biodiversity databases and the need for appropriate biodiversity surrogates. Although these procedures represent alpha diversity patterns successfully, without due consideration of underlying processes and turnover patterns, the long-term persistence of biodiversity within areas identified will not be guaranteed. As land-use changes pose the single most important threat to global biodiversity, the inclusion of land-use data in conservation planning is an essential, but often overlooked component. Current land-uses will expand with growing human populations and expected future land-uses should also be an important component of conservation area selection. This thesis addresses these weaknesses in developing a conservation plan for the Northern Province of South Africa. Incomplete datasets can be addressed by the use of indicator taxa and broad-scale environmental classes. However, these surrogates are not as effective at representing rare and endemic biodiversity features and the specific assessment technique used to test the validity of biodiversity surrogates affects the levels of support found. The inclusion of beta diversity and land-use threats (both current and potential) into conservation area selection highlights shortcomings in more traditional techniques. These forms of data make for more realistic conservation area outputs, however, this comes at an increased cost to land. In a final integrative assessment all areas identified as having high biodiversity value in the preceding analyses are assessed as to the threats they face in order to prioritise these areas for immediate conservation attention. This study addresses many weaknesses in conservation planning techniques, contributing to them becoming real-world conservation tools. In South Africa shortages of conservation resources, as well as land redistribution issues, make conservation planning even more challenging. The need to make these procedures flexible, efficient and realistic is essential. The role of off-reserve conservation areas may help address these difficulties and ensure the persistence of biodiversity in one of the world's most biodiverse regions.

**Keywords:** biodiversity, conservation, reserve selection, surrogacy, turnover, land-use, sustainability, Northern Province

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## Disclaimer

This thesis consists of a series of chapters and appendices that have been prepared for submission to, or publication in, a range of scientific journals. As a result styles may vary between chapters and appendices in the thesis and overlap may occur to secure publishable entities.

# Table of Contents

# CHAPTER 1

## General Introduction

*"Only after the Last Tree has been cut down,*
*Only after the Last River has been poisoned,*
*Only after the Last Fish has been caught,*
*Only then will you find that*
*Money Cannot Be Eaten."*
Cree Indian Prophecy

Biodiversity, the diversity of organisms, their genes and the environment in which they interact, faces large threats mostly in the form of human population growth and associated land transformations (Soulé, 1991; Dale *et al.*, 1994; Sala *et al.*, 2000). The resultant, mostly human-induced, species extinction rates rival mass extinctions of the geological past and threaten not just the natural world, but also the ecological products and services on which we depend (Kunin & Lawton, 1996; Chapin *et al.*, 2000; Pimm & Raven, 2000; Tilman, 2000). The importance of these natural resources and services is irrefutable, however existing global conservation efforts are mostly inadequate (Pressey, 1994a; Lombard, 1995a,b; Rodrigues *et al.*, 1999). Many of the present day conservation areas were proclaimed in an *ad hoc* and opportunistic fashion and include areas with high scenic values, high tourism potential and low potential for other forms of land-use (e.g. agriculture or forestry) (Pringle, 1982; Pressey *et al.*, 1993; Freitag *et al.*, 1996). This form of conservation area selection is highly inefficient, providing a biased representation of regional biodiversity and is less cost effective in the long run (Pressey & Tully, 1994; Rodrigues *et al.*, 1999). These shortcomings have highlighted a need for effective and systematic conservation area selection techniques in order to identify areas essential to biodiversity conservation (Williams, 1998; Margules & Pressey, 2000).

**Conservation area selection techniques**

These techniques run on a database of biodiversity features and the sites or areas in which these features occur (Margules *et al.*, 1988; Margules & Redhead, 1995; Pressey & Logan, 1998). Features can include species, land facets, vegetation types or any other spatial features. Techniques include the traditional hotspots approach where sites with many features (richness hotspots), many rare features (rarity hotspots) and many endemic features (endemicity hotspots) are identified as priority conservation areas (Prendergast *et al.*, 1993; Lombard, 1995b; Williams *et al.*, 1996). Scoring procedures were applied in the 1980's as a conservation area selection technique. Here sites were ordered according to a combined score from a variety of criteria such as diversity, rarity, size and naturalness of the sites and then selected from the top site down until all features are represented a required number of times (Pressey & Nicholls, 1989). Understandably this leads to an overrepresentation of many features, as well as a biased set of sites depending on the criteria used. However, since then there has been much development in the field of conservation area selection.

With the advent of the principles of complementarity, efficiency and flexibility, among others, these selection techniques have become powerful land-use planning tools (Pressey *et al.*, 1993). The principle of complementarity ensures that it is not just the site with the most features that is chosen, but rather the site that contains the most so far unrepresented features. This then helps ensure the principle of efficiency whereby maximum biodiversity features are represented in the minimum number of sites possible. Flexibility implies that for features that do occur in alternate sites, these sites are also highlighted as possible selections to allow for flexibility of choice in land-use planning. This principle of flexibility is related to an additional component of conservation area selection and that is the concept of irreplaceability. The irreplaceability of a site is a measure of how important that site is to the conservation goals within a region. In other words, how would the loss of that site impact on conservation options in the region. The site's irreplaceability depends on the conservation targets set. A site that is totally irreplaceable, for a conservation target of 100% species representation, will be one that contains a species found nowhere else in the region. Irreplaceable sites decrease the flexibility of conservation options within a region (Pressey *et al.*, 1994; Ferrier *et al.*, 2000).

The successful inclusion of these principles and others into heuristic iterative algorithms and optimising linear programming algorithms has made for powerful conservation planning tools (Church *et al.*, 1996; Csuti *et al.*, 1997; Williams, 1998; Margules & Pressey, 2000). Heuristic algorithms proceed in a step-wise fashion selecting sites with the most so far unrepresented features (richness-based algorithms) or the highest number of so far unrepresented rare features (rarity-based algorithms) (Kirkpatrick, 1983; Margules *et al.*, 1988; Pressey & Nicholls, 1989; Bedward *et al.*, 1992; Nicholls & Margules, 1993; Margules *et al.*, 1994a; Freitag *et al.*, 1997; Pressey *et al.*, 1997; van Jaarsveld *et al.*, 1998). Optimising linear programming algorithms utilise what operations researchers call a maximal coverage problem and often find the most optimal solution for representing maximum features in the minimum amount of area, although this optimality comes with a trade-off of computational time required (Church *et al.*, 1996; Csuti *et al.*, 1997). However, these techniques have increased in sophistication, power and speed over recent years.

There are however, several shortcomings associated with this suite of conservation area selection techniques. These include incomplete feature databases, inadequate representation of ecosystem processes, patterns of spatial and temporal feature turnover, shifting anthropogenic threats, and the need to take current and potential development opportunities into account (Balmford *et al.*, 1998; Williams, 1998; Maddock & Du Plessis, 1999; Margules & Pressey, 2000; Nicholls, 1998; Wessels *et al.*, 2000 (see Addendum II); Mace *et al.*, 2000).

*Shortcomings in conservation area selection*

<u>Databases</u>

The aim of conservation area selection techniques is to represent biodiversity within selected sites.

However, the difficulties with sampling the full complexity of biodiversity in order to represent it are often almost insurmountable. Thus the selection of representative minimum-set conservation areas often depends on substitute or surrogate biodiversity data which can be surveyed in a more cost and time efficient manner (Noss, 1990; Vane-Wright *et al.*, 1991; Ryti, 1992; Belbin, 1993; Gaston & Williams, 1993; Pressey, 1994b; Williams & Gaston, 1994a,b; Margules & Redhead, 1995; Pressey & Logan, 1994; Faith & Walker, 1996b; Gaston, 1996a; Williams, 1998). These data include higher taxa, phylogenetic diversity, species richness and broad-scale environmental measures. Areas rich in higher taxa (e.g. families or orders) are assumed to be rich in lower taxa (e.g. species) and therefore contain much biodiversity (Gaston & Williams, 1993; Williams & Gaston, 1994a; Balmford *et al.*, 1996). Data on higher taxa are often more easily obtainable than data on the lower levels. Phylogenetic diversity measures how closely related the species in an assemblage are in evolutionary terms, and thus captures more of the biodiversity than other surrogate measures (Vane-Wright *et al.*, 1991; Faith, 1992; 1994; Williams & Humphries, 1994). These data, however, are very labour intensive to obtain and are, more often than not, unavailable.

Species richness is one of the most common currencies of biodiversity measurement (Heywood, 1994; Gaston & Spicer; 1998). These data are widely and often well collected, especially for some taxa e.g. mammals, birds and vascular plants. It is often the form of data used by conservation area selection techniques, and usually comprises the distribution of species recorded as presence/absences in sites such as grid cells, forest reserves and water catchments. However, species distribution data have many shortcomings. The taxa employed are often poorly known taxonomically and incompletely surveyed with biased survey records for a region. As Polasky *et al.* (2000) point out existing methods for the selection of areas important for species conservation rely on data on the presence or absence of species in various sites. These data are seldom available since not all of the sites have been sampled for all species and therefore the probability of false absences is high. It has therefore been suggested that a useful surrogate or substitute for species richness data could be indicator taxa (Prendergast *et al.*, 1993; Lombard, 1995b; Williams *et al.*, 1996; Flather *et al.*, 1997; Balmford, 1998; Howard *et al.*, 1998; van Jaarsveld *et al.*, 1998). These are taxonomic groups that are well-known taxonomically and well surveyed within the region of interest. The word 'indicator' in this study as in biodiversity indicator, indicator taxon etc. implies a group or taxon used to locate areas of high biodiversity and thus help in conservation planning. It is not used in the sense of indicator species which are employed in assessing environmental quality and human impacts (Caro & Doherty, 1999). It is then assumed that patterns in these indicator groups reflect patterns in other unsurveyed taxa. However, this assumption has seldom been assessed and results are often conflicting as to the validity of indicator taxa as surrogates for biodiversity. Chapter 2 provides a regional assessment of indicator taxa in an effort to test this assumption.

These surrogate measures all have important contributions to make toward the quantification of biodiversity patterns and the identification of areas important to its conservation. However, the assumed relationship between these measures and the underlying biodiversity has seldom been investigated and due to the inadequacy of most biodiversity data will remain difficult to investigate. There are a variety of techniques available for the assessment of the effectiveness of biodiversity surrogates each of which provide different levels of support for the use of surrogates. These techniques include assessments of the degree of overlap and representativeness of conservation areas based on different biodiversity surrogates (Prendergast *et al.*, 1993; Lombard, 1995b; Gaston, 1996b; Flather *et al.*, 1997; Howard *et al.*, 1998; van Jaarsveld *et al.*, 1998). Chapter 3 investigates the impacts these various assessment techniques have on the degree of support offered for the use of indicator taxa as biodiversity surrogates. It is, however, also argued that species comprise just one level of the biodiversity hierarchy and as such are an inadequate representation of the diversity found within nature's hierarchy (Noss, 1990; 1996; Faith & Walker, 1996a; Maddock & Du Plessis, 1999).

A final surrogate for biodiversity is broad-scale biological and environmental data. This form of data includes vegetation types, land facets, land classes and land systems, and because it comprises a higher level of the biodiversity hierarchy is expected to capture much diversity found in lower hierarchical levels (Pressey; 1994b; Pressey & Logan, 1994; Wessels *et al.*, 1999; Fairbanks & Benn, 2000). Chapter 4 provides a regional assessment of the broad-scale biodiversity surrogates of vegetation and landtypes and their success at representing regional species diversity.

Biodiversity processes and feature turnover

A recurrent problem with most existing conservation area selection techniques is that although they may achieve varying levels of success in representing existing biodiversity, they concentrate primarily on biodiversity pattern. Representation of current patterns of species diversity, vegetation types or land classes in conservation areas is only one facet of successful biodiversity representation. This form of representation ignores the dynamic nature of biodiversity features e.g. the movement of individuals, populations, migration, population processes and viability, disturbance regimes, climate change and the ecological interactions between species and their environment within a community (Balmford *et al.*, 1998; Cowling *et al*, 1999). This tends to suggest that a pattern-only based approach towards the identification of conservation areas will not guarantee the long-term maintenance of both biodiversity pattern and the processes responsible for that pattern (Nicholls, 1998; Williams, 1998). Conservation of ecosystem processes that sustain ecosystem structure and function, and evolutionary processes that sustain lineages and generate diversity, are essential for achieving the long-term maintenance of biodiversity in conservation areas (Nicholls, 1998; Cowling *et al.*, 1999; Margules & Pressey, 2000). Very little work has been done on the effects of these processes on the continued representation of biodiversity within selected conservation areas. Some initial work has shown that conservation areas based on biodiversity patterns at one moment in time will not continue to represent that biodiversity some years down the line (Rodrigues *et al.*, 2000).

Another shortcoming of this pattern-based approach is that although it focuses on the representation of diversity, this diversity is mostly alpha diversity. Alpha diversity is the number of species within a homogenous community (Whittaker, 1972; 1977); beta diversity on the other hand is concerned with species turnover or the rate at which species are replaced by others along habitat gradients (Whittaker, 1972). This form of diversity is of crucial importance in conservation area identification, as it provides an indication of feature turnover both in space and in time and is an important determinant of regional species richness patterns. Conventional reserve selection techniques aim to represent all species in a complementary fashion based on a brief snapshot of their distribution patterns. However ignoring the dynamic nature of these patterns, as they change through time and space, may result in conservation areas able to represent current biodiversity patterns, but unable to maintain biodiversity in the long-term (Margules *et al.*, 1994b; Virolainen *et al.*, 1999; Rodrigues *et al.*, 2000). Chapter 5 applies spatial surrogates of biodiversity processes and feature turnover in conservation area selection.

Anthropogenic threats

The basic role of conservation areas is to protect elements of biodiversity from external processes and factors that threaten their existence (Margules & Pressey, 2000). Very few of the existing methods for identifying conservation areas include measures of threat into the selection process (Balmford *et al.*, 1998; Faith & Walker, 1996c; Williams, 1998). Some of these threats are natural and include demographic, genetic and environmental fluctuations and stochasticity, which can be further aggravated by human impacts. Most of the threats facing biodiversity today are anthropogenic in origin and include land development and the associated fragmentation, degradation and land transformation, over-exploitation, artificial species introductions and translocations, and pollution (Lande, 1998). Human population expansion and the development of land results in land-cover changes, mainly due to agriculture and urban development, and present the single most important threat to global biodiversity (Soulé, 1991; Dale *et al.*, 1994; Sala *et al.*, 2000).

Many sites identified by traditional conservation area selection procedures as important to biodiversity conservation may in reality be largely transformed and the features said to exist there may now be extinct (especially in the case of historic data) (Wessels *et al.*, 2000 (see Addendum II)). Or else these areas, being so heavily transformed, may not be able to sustain biodiversity features and processes without intensive and costly management (Baudry, 1993; Di Benedetto *et al.*, 1993; Hobbs, 1993; Freemark, 1995; Allan *et al.*, 1997). Methods therefore need to be developed to identify these areas in order to either avoid them in conservation area selection, or if this is not possible, due to high irreplaceability values of particular sites, then to highlight these areas for immediate conservation (Lombard *et al.*, 1997; Nantel *et al.*, 1998; Wessels *et al.*, 2000 (see Addendum II)). Chapter 5 includes land-cover information into conservation area selection in an effort to address this shortcoming.

Development opportunities

The final shortcoming identified in conservation area selection techniques in the present study is the fact that they do not usually allow for the consideration of future development opportunities and their impacts on biodiversity (Dale *et al.*, 1994; Freemark *et al.*, 1995; White *et al.*, 1997; Pressey, 1998). Sustainable development or "development that meets the needs of the present generation without compromising the needs of future generations" was a phrase made familiar by the World Commission on Environment and Development in 1987. This need to simultaneously address environmental and developmental requirements was highlighted. Thus the integration of conservation and development is essential in order to achieve sustainability now and for future generations.

Chapter 5 provides a useful method for the inclusion of current and past land-uses into conservation planning, it is, however, important to remember that human land-use impacts are not static and will continuously expand as populations and their land-use needs evolve. This has important implications for conservation as it increases costs, decreases conservation options and increases the amount of conflict between the various forms of land-use and conservation. It is therefore essential that natural areas with high potential to become transformed by other land-uses be identified as early as possible in order to identify areas where future conflict between such potential developments and biodiversity are likely. A conservation area selection technique which avoids areas that are currently largely transformed, identifies areas crucial to biodiversity conservation and needing immediate intervention (see Addendum II) and also identifies untransformed areas that are suitable for future developments will hopefully contribute to the persistence of regional biodiversity (Pressey *et al.*, 1996; Williams, 1998). A better understanding of the current and future threats facing biodiversity will allow for more effective trade-offs between achieving biodiversity conservation goals and realising development opportunities (Faith, 1995; Faith & Walker, 1996d), as well as a more efficient immediate allocation of limited conservation resources towards areas most at risk (Margules & Pressey, 2000).

Pressey (1997) and Cowling *et al.* (1999) highlight the fact that many of the existing conservation area selection techniques say nothing about the relative needs of areas selected for protection. Funding and resource shortages dictate that although a large number of areas may be identified as important for the representation of biodiversity, only a fraction of them can be protected in the near future. In order to maximise the retention of biodiversity features within a region, one must minimise the extent to which the original representation goals are compromised by habitat loss while the conservation area network is developing (a process that can take decades) (Cowling *et al.*, 1999). It is therefore crucial to identify areas of high conservation value or urgency within this selected set of areas. These are areas with a high biodiversity value, as well as a high threat or vulnerability value (Faith & Walker, 1996c; Pressey *et al.*, 1996; Pressey, 1997; Pressey, 1998; Cowling *et al.*, 1999).

Much work has been done on measuring biodiversity values of areas (Williams *et al.*, 1996; Williams, 1998; van Jaarsveld *et al.*, 1998) and includes measures of biodiversity pattern (Chapters 2, 3

& 4) and biodiversity processes and turnover (Chapter 5) (Pressey, 1994; Pressey *et al.*, 1994; Noss 1996; Balmford *et al.*, 1998; Pressey, 1998; Maddock & du Plessis, 1999; Ferrier *et al.*, 2000; Rodrigues *et al.*, 2000). However, there is a considerable need for work on the inclusion of threat or vulnerability values of areas into conservation planning. (Addendum II; Faith & Walker, 1996c; Pressey *et al.* 1996; Williams, 1998).

Therefore as a final concluding assessment for this thesis, Chapter 6 brings together all the results and outputs of the previous analyses in order to put together a regional multicriteria-based conservation plan for the Northern Province. Using these results it determines the biodiversity values of areas in the province based on measures of biodiversity pattern, process and turnover. It then proceeds to provide threat values for these areas by identifying currently transformed areas as well as untransformed areas suitable for future land-uses in an attempt to include both current and future land-use patterns into conservation area selection. This is done in order to evaluate the threats these existing and future land-uses pose for regional biodiversity and conservation planning. Through the incorporation of threat values into conservation planning Chapter 6 provides an assessment of the relative need of areas with high biodiversity value for immediate conservation action.

**Aims**

This study therefore aims to address these shortcomings in existing conservation area selection techniques by:

i) Assessing the assumed relationship between indicator taxa and the non-target species they are meant to represent in the identification of conservation areas (Chapter 2).

ii) Evaluating the effects of the different methods of assessment, used in Chapter 2 and other studies, on the validity of indicator taxa as biodiversity surrogates (Chapter 3).

iii) Investigating the value of broad-scale environmental classes as surrogates for regional biodiversity (Chapter 4).

iv) Determining the impact of the inclusion of feature turnover and measures of beta diversity into conservation area selection (Chapter 5).

v) Assessing the value of land-use data in conservation area selection in an effort to minimise threat in conservation areas and highlight areas of potential conflict (Chapter 5).

vi) Identifying currently untransformed areas suitable for alternate land-uses in an effort to identify future land-use threats to biodiversity (Chapter 6).

vii) Finally, through the use of all methods investigated in previous chapters of this thesis, areas with high biodiversity value will be identified. These areas will then be investigated as to their current and potential threat values in an effort to determine their relative conservation urgency (Chapter 6).

**Study area**

The study area comprises the Northern Province of South Africa, which lies between the lines of latitude 22° 00' S to 24° 00' S, and from 26° 00' E to 32° 00' E. One of the nine provinces of South Africa, it occupies about 10% (122305 $km^2$) of the country and lies at the northeastern tip of South Africa. It borders on the countries of Mozambique to the east, Botswana to the west and Zimbabwe to the north. Its southern boundary is made up of three South African Provinces, the Mpumalanga, Gauteng and North West Provinces (Figure 1).

The province includes the northern end of the Drakensberg escarpment which separates the low-lying, warm and more humid Lowveld region in the east from the higher lying, drier and cooler Bushveld plateau region in the west (Figure 1). The Limpopo River forms the northern and northeastern boundary of the province where it borders on the neighbouring states of Botswana and Zimbabwe. This Limpopo River valley is separated from the Lowveld and central Bushveld plateau by the Soutpansberg and Blouberg mountain ranges (Figure 1). These mountain ranges are of ecological, economic and social importance. Steep environmental gradients imply a diversity of species and habitat, which in turn implies a high conservation value. In addition, spectacular scenery provides good opportunities for conservation based tourism. The area also has a high potential for forestry and agriculture in places (Butt *et al.*, 1994). An expanding and generally poor human population is an additional feature of these ranges. The Waterberg mountain range falls within the central Bushveld plateau region and together with the escarpment encircles the Springbok flats, a clay substrate basin within the Bushveld plateau with a long history of dryland cultivation (Figure 1).

*Climate and vegetation*

The climate of the Northern Province is primarily a lowveld dry tropical and dry subtropical one. Rainfall is low, highly variable and seasonal with a distinct dry season during the winter months. Humidity is low and day temperatures are high even in the winter. However, the mountainous areas of the escarpment, Waterberg, Blouberg and Soutpansberg ranges provide marked climatic gradients due to the influence of extreme physiographic relief (Fairbanks, 1997). These mountainous areas have a marked moist tropical to moist subtropical climate with an average to high rainfall that is variable and distinctly seasonal. Here winter minimum temperatures can be low and frost often occurs in valley areas, while humidity can be very high in summer.

The province consists primarily of the savanna biome, with small areas on the escarpment covered by grasslands and forest biomes (Low & Rebelo, 1996). The savanna biome, also referred to as the woodland biome, has a grassland understorey with a woody upperstorey of trees and tall shrubs. Tree cover varies from sparse to almost closed-canopy cover (Rutherford & Westfall, 1986). Grasses are the dominant vegetation in the grasslands biome, with geophytes and herbs also well represented (Low & Rebelo, 1996).

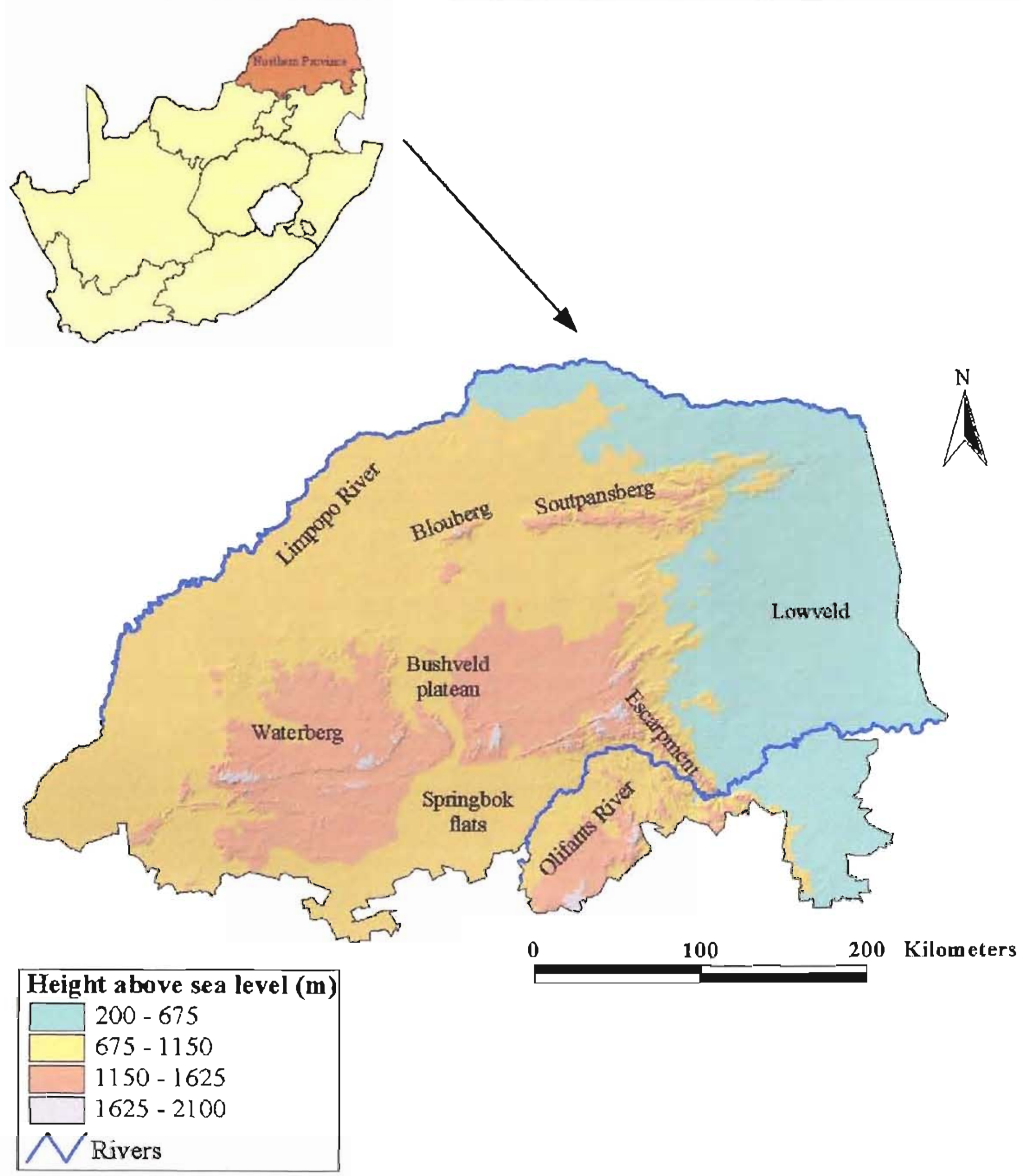


**Figure 1:** Map of the study area of the Northern Province, showing topography, major rivers, mountain ranges and broad biogeographical zones.

High summer rainfall, frequent fires, frost and grazing are responsible for the exclusion of trees and shrubs and thus the maintenance of these grasslands (Low & Rebelo, 1996). Tree cover in the forest biome is almost continuous and includes mostly evergreen species (Rutherford & Westfall, 1986). Below the canopy vegetation is multi-layered and there is thick leaf litter and little ground vegetation. Forests occur in frost-free regions with high rainfall and infrequent fires. There are 15 recognised vegetation types that fall within the Northern Province of which 12 are within the savanna biome, two within the grasslands biome and one within the forest biome (Table 1; Figure 2).

*Current land-uses*

The principle urban centres include Pietersburg and Louis Trichardt in the Lowveld, and Tzaneen on the Escarpment (Figure 3). The province includes extensive areas of arable land and as a result 14% of the province has been transformed by cultivation (Table 2; Figure 4). However due to the relatively low rainfall in most parts of the province, dryland cultivation at a commercial scale, which makes up two percent of the total cultivation in the province, is limited to the escarpment, mountainous regions and Springbok Flats, where it is a viable option. In the rest of the province rainfed agriculture is not possible at a commercial scale and is limited to temporary and subsistence level cultivation, making up 38 and 48% of the total area under cultivation in the province respectively. Other areas under cultivation require irrigation. Because of the aridity of the province this form of cultivation is very limited making up three percent of the total cultivation at a commercial level and eight percent at a temporary level (Table 2; Figure 4).

Urbanisation (1.6%) and forestry plantations (0.8%) account for the remaining land transformations (Thompson, 1996; Fairbanks *et al.*, 2000). Therefore the study area has not been excessively degraded and transformed since 73% is still covered by natural vegetation and 11.36% is under formal protection in provincial and national protected areas (Figure 4). This large amount of protected area coverage is due mostly to the Kruger National Park, a National Park with an area of 19600 $km^2$ of which just over 50% falls within the Northern Province (Figure 3). There are about 50 other formally protected provincial and national parks in the Northern Province (Figure 3) (DEAT, 1996).

*Potential land-uses*

Although current land-use impacts on the province have been of a restricted nature, taking into consideration South Africa's expanding human population and the likely increased demands on land and resources, it can be expected that land-use impacts will increase. The ability to identify currently untransformed areas where these land-uses will be expected to impact is of critical importance for the maintenance and future protection of biodiversity. Afforestation, cultivation and mining are considered to be major land-uses that threatened biodiversity.

**Table 1**: Vegetation types, the biomes in which they occur and the extent of each within the Northern Province. (Low & Rebelo, 1996)

| **Vegetation types** | **Biomes** | **Area ($km^2$)** | **%** |
|---|---|---|---|
| Afromontane Forest | Forest | 242.11 | 0.20 |
| Clay Thorn Bushveld | Savanna | 8328.83 | 6.78 |
| Kalahari Plains Thorn Bushveld | Savanna | 110.15 | 0.09 |
| Lebombo Arid Mountain Bushveld | Savanna | 438.44 | 0.36 |
| Mixed Bushveld | Savanna | 35065.86 | 28.54 |
| Mixed Lowveld Bushveld | Savanna | 9327.76 | 7.59 |
| Moist Sandy Highveld Grassland | Grassland | 47.15 | 0.04 |
| Mopane Bushveld | Savanna | 20532.01 | 16.71 |
| Mopane Shrubveld | Savanna | 2590.80 | 2.11 |
| North-eastern Mountain Grassland | Grassland | 3800.49 | 3.09 |
| Sour Lowveld Bushveld | Savanna | 7788.47 | 6.34 |
| Soutpansberg Arid Mountain Bushveld | Savanna | 4788.61 | 3.90 |
| Sweet Bushveld | Savanna | 17212.01 | 14.01 |
| Sweet Lowveld Bushveld | Savanna | 250.01 | 0.20 |
| Waterberg Moist Mountain Bushveld | Savanna | 12356.45 | 10.06 |

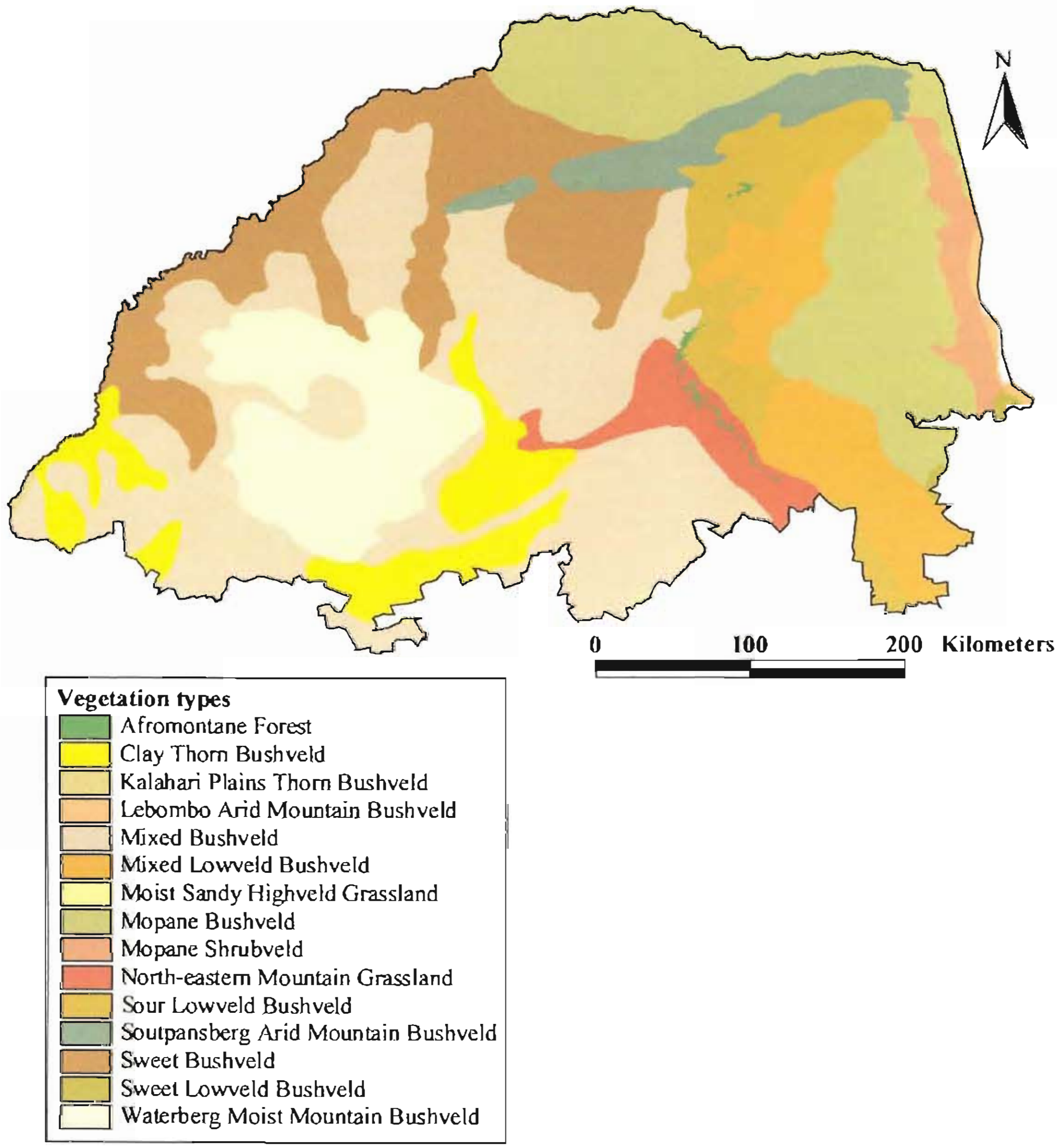


**Figure 2:** Vegetation types of the Northern Province as defined by Low and Rebelo (1996).

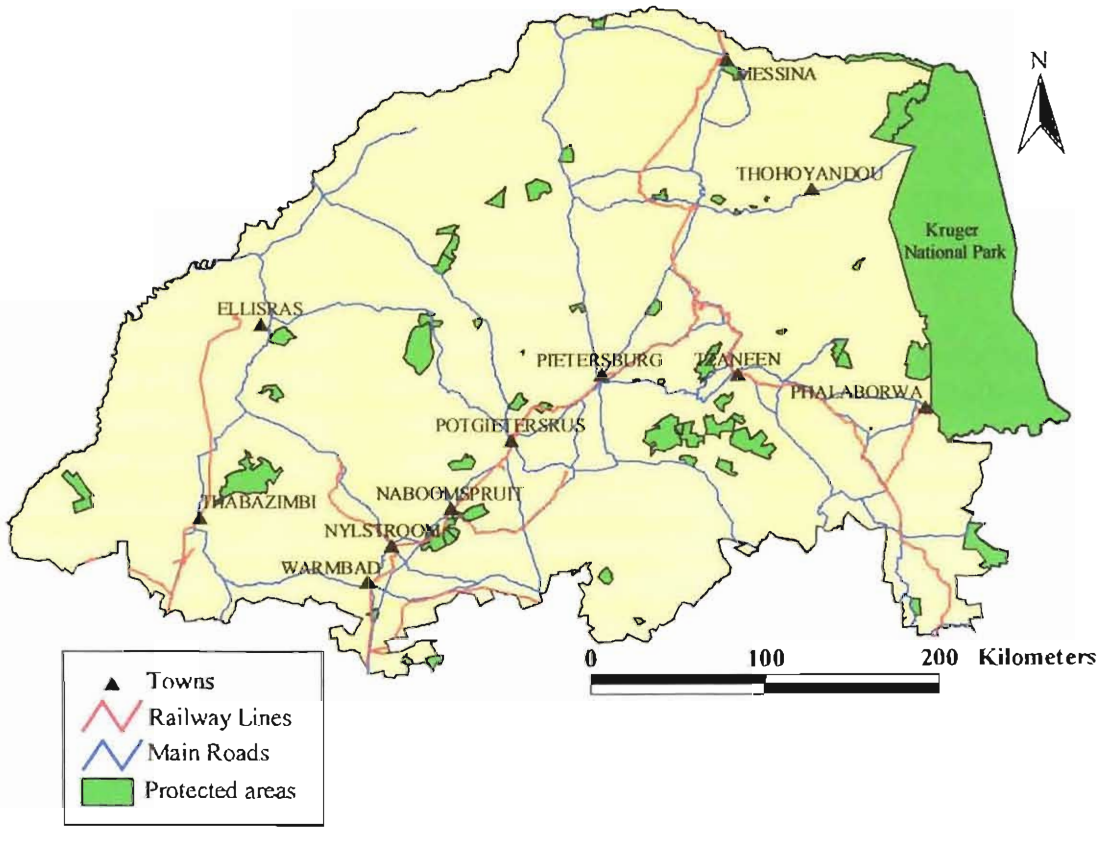


**Figure 3:** Map of the study area showing towns, transport routes and national and provincial protected areas.

**Table 2:** Areas of land-cover categories in the Northern Province of South Africa, illustrating the percentage coverage of each category within the province (extracted from Fairbanks *et al.*, 2000).

| **Land-cover category** | **Area (km$^2$)** | **%** |
|---|---|---|
| Barren rock | 65.96 | 0.05 |
| Cultivated: permanent - commercial dryland | 411.37 | 0.34 |
| Cultivated: permanent - commercial irrigated | 587.04 | 0.48 |
| Cultivated: permanent - commercial sugarcane | 0.00 | 0.00 |
| Cultivated: temporary - commercial dryland | 6599.67 | 5.39 |
| Cultivated: temporary - commercial irrigated | 1606.17 | 1.31 |
| Cultivated: temporary - semi-commercial / subsistence dryland | 7999.27 | 6.53 |
| Degraded: herbland | 0.00 | 0.00 |
| Degraded: forest and woodland | 6476.89 | 5.29 |
| Degraded: shrubland and low fynbos | 0.00 | 0.00 |
| Degraded: thicket and bushland (etc) | 5515.35 | 4.51 |
| Degraded: unimproved grassland | 150.43 | 0.12 |
| Dongas and sheet erosion scars | 77.77 | 0.06 |
| Forest | 376.50 | 0.31 |
| Forest and Woodland | 40165.35 | 32.81 |
| Forest plantations | 992.36 | 0.81 |
| Herbland | 0.00 | 0.00 |
| Improved grassland | 3.89 | 0.00 |
| Mines & quarries | 145.13 | 0.12 |
| Shrubland and low Fynbos | 29.43 | 0.02 |
| Thicket and bushland (etc) | 47792.13 | 39.04 |
| Unimproved grassland | 1359.51 | 1.11 |
| Urban / built-up land: commercial | 14.60 | 0.01 |
| Urban / built-up land: industrial / transport | 27.75 | 0.02 |
| Urban / built-up land: residential | 1727.05 | 1.41 |
| Urban / built-up land: residential (small holdings: bushland) | 60.02 | 0.05 |
| Urban / built-up land: residential (small holdings: grassland) | 0.00 | 0.00 |
| Urban / built-up land: residential (small holdings: shrubland) | 0.05 | 0.00 |
| Urban / built-up land: residential (small holdings: woodland) | 84.34 | 0.07 |
| Waterbodies | 128.16 | 0.10 |
| Wetlands | 17.60 | 0.01 |

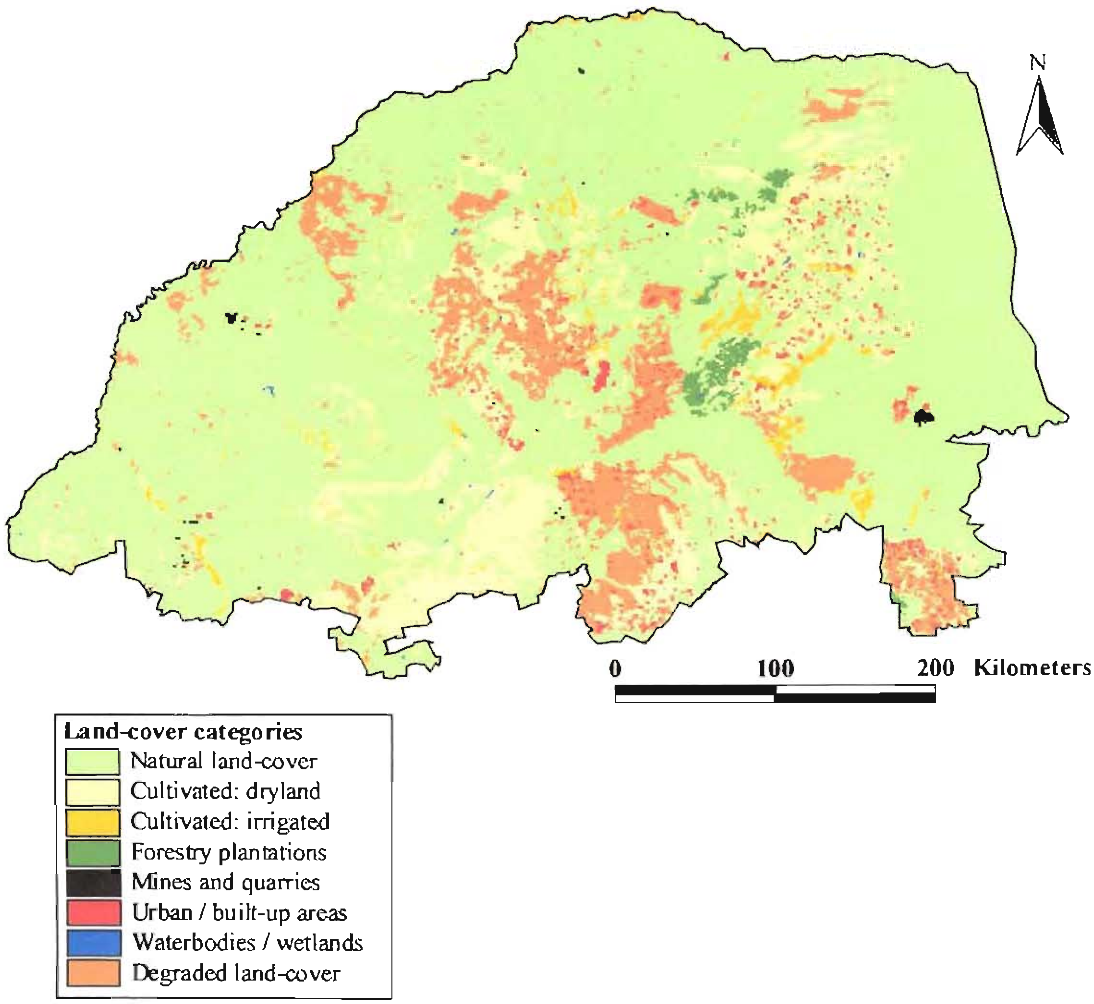


**Figure 4:** Land-cover categories of the Northern Province, illustrating natural, transformed and degraded land-cover.

The Northern Province, being a low rainfall area, does not contain much potential for further afforestation or dryland cultivation, except through specialised species (Fairbanks, 1997).

The Northern Province, although not one of the most important mining provinces in South Africa, is still particularly dependent on the contribution of the mining sector. The province's export and local mineral sales made up 10% of South Africa's sales for 1995 (Wilson & Anhaeusser, 1998). There are however, several mineral and dimension stone fields, provinces, as well as deposits within the province that still remain unexploited (Figure 5) (Wilson & Anhaeusser, 1998). The effects of the potential mining of these areas on surrounding environments and biodiversity should be carefully considered.

Finally, one of the most widespread forms of alteration of natural habitats and landscapes over the last century has been the construction and maintenance of roads (Trombulak & Frissell, 2000). These networks cover 0.9% of Britain and 1.0% of the USA (Forman & Alexander, 1998), however the road-effect zone, the area over which significant ecological effects extend outward from the road, is usually much wider than the road and roadside. Some evidence on the size of the road-effect zone is available from studies in Europe and North America. Reijnen *et al.* (1995) estimated that road-effect zones cover between 12-20% of The Netherlands, while Forman (2000) illustrated that 19% of the USA is affected ecologically by roads and associated traffic.

Therefore the potential impacts of expanding areas under cultivation or forestry plantations, mining developments, as well as the effects of road networks on biodiversity within the province are an essential component of real-world conservation planning.

**Databases**

Several forms of data were employed in this study including species distribution data for a variety of taxa, broad-scale environmental data (e.g. vegetation types), current land–cover data and various potential land-use datasets.

*Species distribution data*

Species distribution data for the Northern Province, as well as the rest of South Africa, are only available at a quarter degree grid cell resolution. These 15' x 15' cells measure approximately 700 km$^2$ in the province. Information on species presence within these grid cells (n = 215) were collated for birds (Aves), butterflies (Lepidoptera: superfamilies Hesperioidea, Papilionoidea), buprestid beetles (Buprestidae), scarab beetles (Scarabaeinae), Termites (Isoptera), mammals (Mammalia), Neuropterans (Myrmeleontidae) and vascular plants (Plantae) (Table 3). Information on avian distribution was collated from the South African Bird Atlas Project (Harrison, 1992; Harrison *et al.*, 1997).

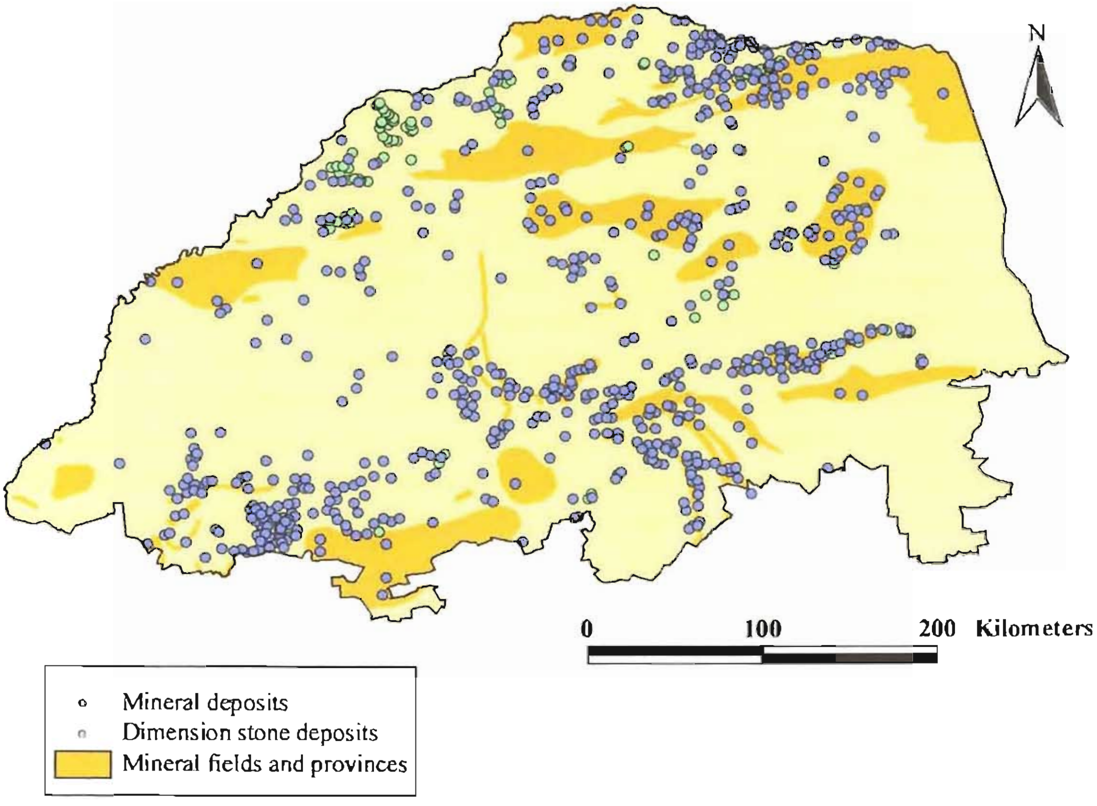


**Figure 5:** Mineral fields, provinces and deposits as well as dimension stone deposits in the Northern Province.

The presence/absence of 574 avian species, which comprise 60% of the bird diversity recorded in the Southern African sub-region (South Africa, Lesotho, Swaziland, Namibia, Botswana, Zimbabwe and southern Mozambique), was recorded from 1980 – 1992. This is the only true presence/absence species database available for South Africa and thus results based on other databases should be treated with caution due to the prevalence of false absences in what are actually presence only databases.

Mammal distribution data were based on primary data collections and species lists of the National Flagship Institute (formerly the Transvaal Museum), United States National Museum African Mammal Collection, National Parks Board (specifically the Kruger National Park), the South African Defence Force, KaNqwane Parks, Rautenbach (1982) and other published and unpublished records (Freitag *et al.*, 1996). These data varied in resolution from point localities to grid cell data of varying resolutions and were therefore generalised to quarter degree grid cells, a grid size determined by the coarsest data resolution. These taxa are all well surveyed within the study area and reflect little survey bias (Harrison, 1992; Freitag & van Jaarsveld, 1995; Freitag *et al.*, 1998)

An extensive butterfly distribution data set as well as buprestid, scarab, termite and neuropteran data sets were collated for South Africa from National Flagship museum records. Sampling localities were transformed to 15' x 15' grid cells with the aid of a gazetteer, resulting in unique distribution records for 613 butterfly, 247 buprestid, 218 scarab, 16 termite and 22 neuropteran species in South Africa (Freitag & Mansell, 1997; Hull *et al.*, 1998; Muller, 1999; Koch *et al.*, 2000). Much work has been done on the systematics and distribution of South African Lepidopterans and only a few remain undescribed (<5%, Owen, 1971). However, the rest of the invertebrate taxa, as is the case for many distribution databases worldwide, are poorly known taxonomically and have biased survey records (Freitag & Mansell, 1997; Hull *et al.*, 1998; Muller, 1999; Koch *et al.*, 2000); their use was therefore limited within this study.

The National Botanical Institute collated higher plant species distribution records at the quarter degree grid cell resolution. These records include 42055 unique distribution records for 5711 species. However a data set of this size sets unattainable formal conservation goals, requiring over 50% of the study area to represent all species only once (Chapter 4). Therefore only endemic plant species (species that were not recorded outside of the former Transvaal Province) were included in the analyses. It is important to note that throughout this study species distribution data, as well as many other forms of spatial data, were collated at a quarter degree grid cell resolution, an area of approximately 700 $km^2$. It has been demonstrated that the spatial scale of biological data will affect conservation planning outputs, as well as evaluations of congruency and prediction accuracy of indicator groups (Pearson & Carroll, 1999; Schwartz, 1999). However, this is the best available data for the study region and is still useful in illustrating basic trends and principles. It is important to remember that these analyses highlight large areas of conservation importance, which can then be investigated at a local scale.

**Table 3:** Species distribution data

| Taxon | Unique records | Unique species | Rare species | Endemic species | Grids surveyed | Survey date |
|---|---|---|---|---|---|---|
| *Well known taxa* | | | | | | |
| Birds (Aves) | 49089 | 574 | 141 | 63 | 214 (99%) | 1980-92 |
| Butterflies (Hesperioidea & Papilionoidea) | 2062 | 328 | 79 | 4 | 84 (39.1%) | 1905-80 |
| Mammals (Mammalia) | 5218 | 214 | 56 | 1 | 183 (85.1%) | 1980-95 |
| Endemic vascular plants (Plantae) | 2694 | 472 | 125 | 472 | 190 (88.4%) | 1900-96 |
| ***Combined*** | ***59063*** | ***1588*** | ***353*** | ***540*** | ***215 (100%)*** | |
| *Less well known taxa* | | | | | | |
| Buprestid beetles (Buprestidae) | 977 | 247 | | | 119 (55%) | 1900-96 |
| Scarab beetles (Scarabaeinae) | 1372 | 218 | | | 124 (58%) | 1900-92 |
| Termites (Isoptera) | 464 | 16 | | | 160 (74%) | 1972-80 |
| Neuropterans (Myrmeleontidae) | 126 | 22 | | | 41 (19%) | 1900-96 |
| ***Combined*** | ***2939*** | ***503*** | | | ***194 (90%)*** | |
| ***Total combined*** | ***61975*** | ***2091*** | | | ***215 (100%)*** | |

*Vegetation data*

Shortcomings with species distribution data as a useful measure of biodiversity have led to a shift in the focus for conservation. This has resulted in recommendations towards a more holistic approach of protecting biodiversity in the aggregate, the so-called 'coarse-filter' approach (Noss, 1990; 1996). The goal of coarse-filter conservation is to preserve all or most species in a region by protecting sufficient (>20000 ha) samples of every plant community type (Scott *et al.*, 1993). Other hierarchical methods have included species assemblages, land facets, or landscapes (Pressey 1994b; Pressey & Logan, 1994; Wessels *et al.*, 1999; Fairbanks & Benn, 2000).

At a national scale South Africa has a few databases of broader surrogates for biodiversity, including Acocks' Veld Types (Acocks, 1988) and the more recent Vegetation of South Africa, Lesotho and Swaziland (Low & Rebelo, 1996; McDonald, 1997). Acocks (1988) defined biological resources from a purely agricultural potential perspective, while Low and Rebelo (1996) looked at the definition of these resources from a management and potential use angle. These vegetation units were defined as having, "... similar vegetation structure, sharing important plant species, and having similar ecological processes". Thus, these are units that would have potentially occurred today, were it not for all the major human-made transformations e.g. agriculture and urbanisation. Therefore the Low and Rebelo (1996) vegetation map contains significant potential for acting as a broad scale surrogate of South African biodiversity and for identifying land important to biodiversity conservation and was employed in the present study. The vegetation types within the study region have already been described in Table 1.

In a recent study on the threat status of the vegetation types of South Africa (see Addendum I), four of the vegetation types found within the Northern Province (Kalahari Plains Thorn Bushveld, Clay Thorn Bushveld, Mixed Bushveld and Sour Lowveld Bushveld) fell within the top 20 most threatened vegetation types within South Africa. This is due to a combination of large transformed and degraded areas and few protected areas within the vegetation type.

*Environmental data*

In the assessments requiring environmental data, the factors and processes that have been hypothesised to account for spatial patterns of species diversity are climatic extremes, climatic stability, productivity, and habitat heterogeneity (Brown, 1995; Wickham *et al.*, 1997). Data were compiled from existing sources to represent these factors (Table 4), including interpolated weather stations (Schulze, 1998) and topographic contours (SA Surveyor General, 1993a) mapped in a geographic information system (GIS; ESRI 1998) using Albers equal area projection. This GIS database had a grid cell resolution of 1km x 1km, which was determined by the cell size of existing rasterised data sets and a cell size that could be used in future analyses.

*Land-cover data*

Current land-cover data

The recent advent of the National Land-cover database (NLC) has allowed for national level assessments of current land-cover in South Africa. This national database was derived using manual photo-interpretation techniques from a series of 1:250000 scale geo-rectified hardcopy satellite imagery maps, based on seasonally standardised, single date Landsat Thematic Mapper (TM) satellite imagery captured principally during the period 1994-95 (Fairbanks & Thompson, 1996). It provides the first single standardised database of current land-cover information for the whole of South Africa, Lesotho and Swaziland (Fairbanks *et al.*, 2000).

For the purpose of future analyses in the present study the 31 land-cover classes (Table 2) were reclassified into three categories: natural, degraded and transformed land-cover (Table 5; Wessels *et al.*, 2000 (see Addendum II)). Natural land-cover included all untransformed vegetation, e.g. forest, woodland, thicket and grassland. The degraded land-cover category was dominated by degraded classes of land-cover. These areas have a very low vegetation cover in comparison with the surrounding natural vegetation cover and were typically associated with rural population centres and subsistence level farming, where fuel-wood removal, over-grazing and subsequent soil erosion were excessive (Thompson, 1996). Grazed areas are not included in this degraded category, unless they are severely over-grazed. In general it can be assumed that all areas of remaining natural vegetation are rangelands used for either domestic or wild livestock grazing. The transformed category consisted of areas where the structure and species composition were completely or almost completely altered which includes all areas under crop cultivation, forestry plantations, urbanised areas, and mines/quarries.

Potential land-cover data

Potential land-cover data were obtained from multiple sources. Potential afforestation was determined by bioclimatic prediction (BIOCLIM) and fuzzy sets logic modeling (Fairbanks, 1997; Fairbanks & Smith, 1995) based on soil information and bioclimatic parameters (e.g. growth days and growth temperature). These variables were provided by Centre for Computing and Water Research (University of Natal) and the ARC - Institute for Soil, Climate and Water. Suitability for agriculture was calculated for both rain-fed and irrigated cultivation by the Institute for Soil, Climate and Water, using data on soil patterns, rainfall, slope and water availability (Schoeman *et al.*, 1986; Smith, 1998).

In the past, suitability mapping was based on Boolean operations, regression models and expert estimates for classifying areas of land. An area was tested on its attribute values as to whether it fell within each set or not, and any entity not matching all criteria was rejected. However, this method assumes that real world criteria can be modelled as discrete entities with exact attributes, and in reality most environmental questions are more complex than this.

**Table 4:** Codes and definitions of environmental variables used.

| Code | Definition |
| --- | --- |
| Topography | |
| DEMMEAN | Elevation (m) |
| DEMSTD | Elevation heterogeneity (std. Deviation) |
| Climate | |
| GDMEAN | Number of days per annum on which sufficient water is available for plant growth |
| MAP | Mean annual precipitation (mm) |
| GTMEAN | Annual mean of the monthly mean temperature (°C) weighted by the monthly GD |
| NGTMEAN | Mean temperature (°C) during negative water balance |
| MAT | Mean annual temperature (°C) |
| MAXMNTHMN | Mean temperature of the hottest month, usually January (°C) |
| MINMNTHMN | Mean temperature of the coldest month, usually July (°C) |
| EVANNMN | Total annual pan evapotranspiration (mm) |
| PSEAS_MN | Precipitation seasonality from the difference between the January and July means |
| TSEAS_MN | Temperature seasonality from the difference between the January and July means |
| MXSEAS_MN | Maximum temperature seasonality from the difference between January and July |

**Table 5:** Land-cover classes reclassified into broad categories (after Wessels *et al.*, 2000 (see Addendum II)).

| Transformation category | % Area occupied in Northern Province | Land-cover class |
|---|---|---|
| Natural land-cover | 73.36 | Wetlands, grassland, shrubland, bushland, thicket, woodland, forest |
| Degraded land-cover | 10.09 | Degraded land, erosion scars, waterbodies |
| Transformed land-cover | 16.55 | Cultivated lands, urban/built-up areas, mines and quarries, forestry plantations |

In addition, because spatial variation is not directly measurable in its entirety but is reconstructed from point data, the resulting input attributes will have errors, this is especially a problem for attributes with values near the boundaries of the sets. The replacement of Boolean sets with fuzzy sets (or continous classes) replaces the finite boundary of the Boolean set with a gradual transition zone, and allows for partial set membership. This then prevents the exclusion of attributes with values just outside the class boundaires. As Fairbanks (1997) points out the use of strict Boolean algebra with simple TRUE/FALSE logic is inappropriate for land suitability evaluation, because of the continuous nature of environmental data and the inexactness of formulating queries.

The road-effect zone for South Africa was determined using a similar method to that used by Stoms (2000) in which the spatial extent of road-effects (road-effect zone) can be used as an ecological indicator that directly represents impacts on biodiversity. The affected distances were estimated in a hierarchical fashion from the reviews mentioned above, as well as from local studies (Milton & Macdonald, 1988). National routes and freeways were assumed to affect biodiversity for a greater distance from the roadway (1km on each side) than farm roads (100 m; Table 6). Road segments from the South African Surveyor General (1993b) 1:500000 scale map series files (SA Surveyor General, 1993b) were buffered in a standard geographic information system operation to the distance related to its class (Figure 6). Although the roads in protected areas do have an impact on biodiversity within these areas, they were excluded from this analysis as by and large protected areas overwhelmingly contribute to biodiversity conservation.

**Geographic Information Systems (GIS) analysis**

The species distribution data collated at a quarter degree grid cell resolution was the coarsest resolution data used within the study and therefore determined the resolution of the remaining data. Thus the vegetation, land-cover and environmental data were overlaid with the 15' x 15' grid (Figure 7). An aggregated mean statistic was recorded for each grid cell for the vegetation, environmental and topographical features found within that grid cell. The extent of current and potential land-cover classes, as well as national and provincial protected areas within each grid cell was calculated using ArcInfo. All GIS analyses were conducted in ArcView and ArcInfo (ESRI, 1998) in Albers equal area projection, with Spheroid Clarke 1880 and using the parameters of reference longitude 24° 00' 00" E and standard parallels of -18° 00' 00"S and -32° 00' 00" S.

**Conservation area selection**

Traditional methods of conservation area selection comprising identification of hotspots of species richness and rarity were used for some aspects of this study (Williams, 1998). However, complementarity-based iterative algorithms were the chief conservation area selection tool employed (Nicholls & Margules, 1993).

**Table 6:** Buffer widths assigned to road classes for calculating road-effect zone (after Stoms 2000).

| South African Surveyor General Description | Buffer width (m) |
|---|---|
| National route | 1000 |
| Freeway | 1000 |
| Arterial | 500 |
| Main | 250 |
| Secondary (connecting and magisterial roads) | 100 |
| Other (rural road) | 50 |
| Vehicular trail (4 wheel drive route) | 25 |

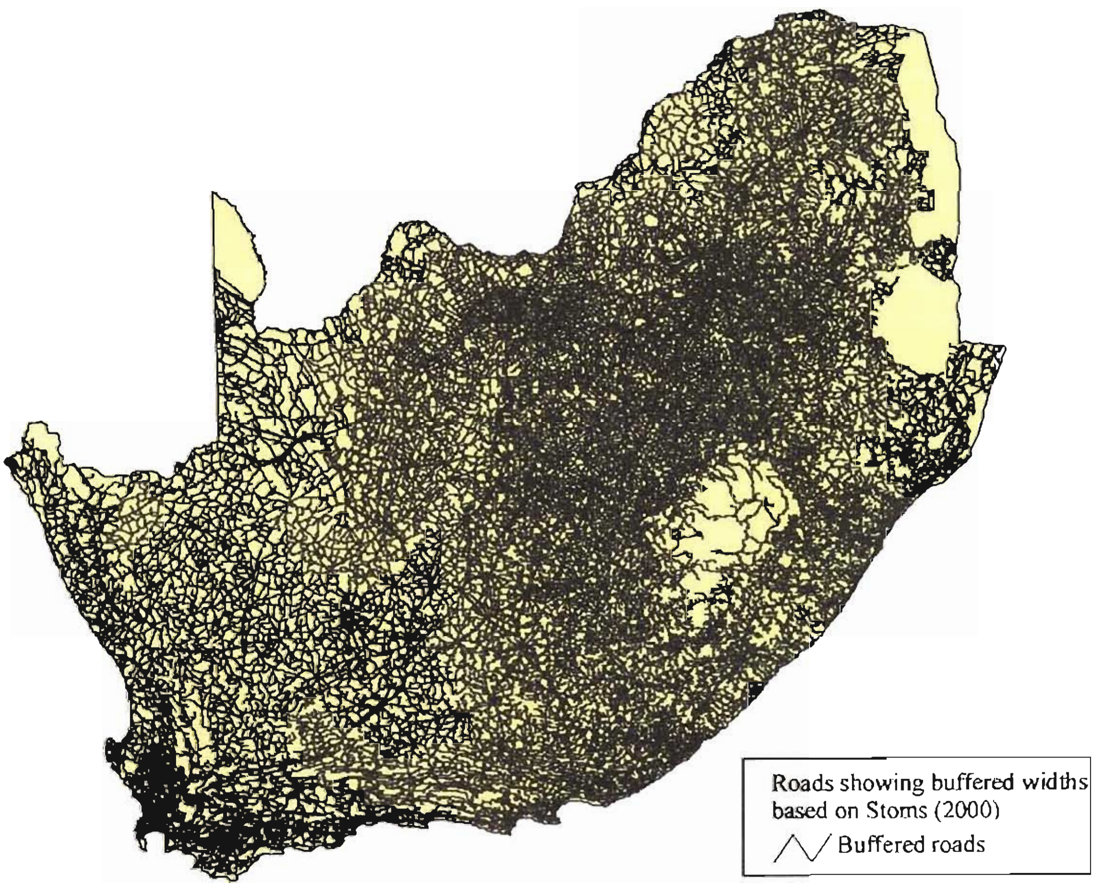


**Figure 6**: Buffered road network of South Africa.

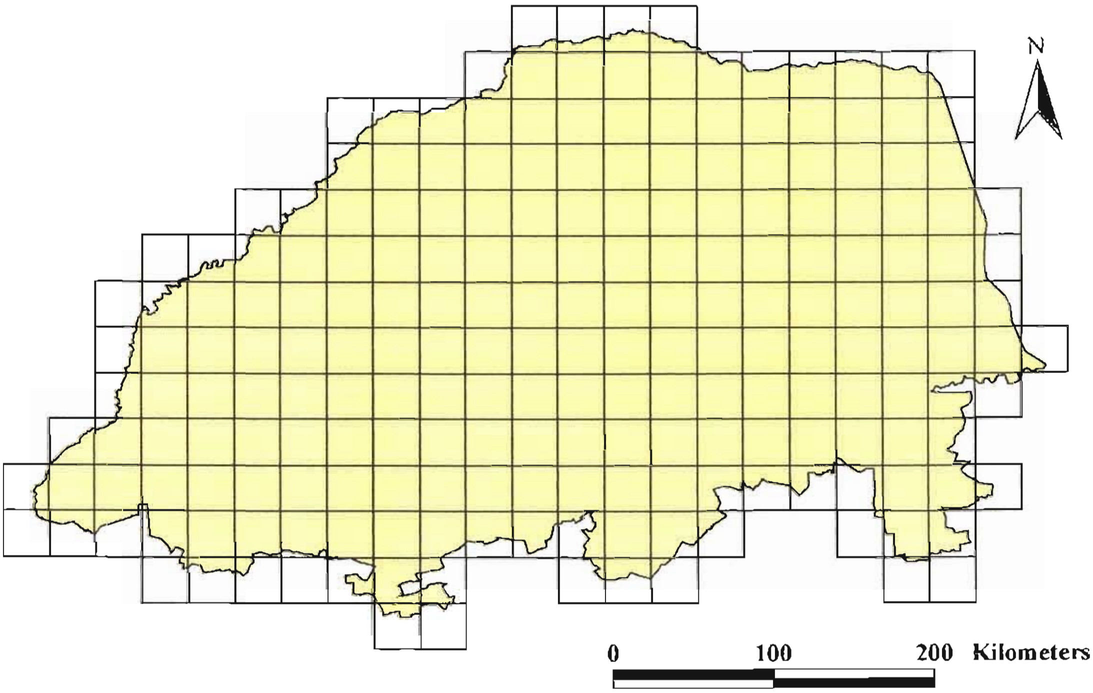


**Figure 7:** Quarter degree grid cells (n = 215) of the Northern Province.

Basic richness- and rarity-based algorithms were programmed and used in many of the analyses. These algorithms were adapted and reprogrammed to meet the requirements of later analyses. An iterative algorithm able to represent a specified percentage of broad-scale surrogate classes was programmed and used in Chapters 4 and 6. Additional adaptations are described in more detail in Chapters 5 and 6 and Addendum II, and include reprogramming for the incorporation of land-use information and beta diversity into conservation area selection. In most cases a 25 to 50% level of preselection was employed. This implies that any biodiversity feature occurring in a site more than 25 to 50% protected is assumed to be already represented and is excluded from future selection.

The terms reserve network, conservation area, priority conservation area and protected area all refer to existing or identified sites for the conservation of biodiversity. These areas include existing formal protected areas (IUCN categories I and II) as well as areas identified as important to biodiversity conservation by this study. The areas identified can then be formally protected or, in the case of land and budgetary constraints, rely on some form of off-reserve management (Pressey & Logan, 1997).

**References**

Acocks, P.H. (1988). Veldtypes of South Africa, 3rd Edition. *Memoirs of the Botanical Survey of South Africa* 57, 1-146.

Allan, D.G., Harrison, J.A., Navarro, R.A., van Wilgen, B.W. & Thompson, M.W. (1997). The impact of commercial afforestation on bird populations in Mpumalanga Province, South Africa - insights from bird-atlas data. *Biological Conservation* 79, 173-185.

Balmford, A. (1998). On hotspots and the use of indicators for reserve selection. *Trends in Ecology & Evolution* 13, 409.

Balmford, A., Jayasuriya, A.H.M. & Green, M.J.B. (1996). Using higher taxon richness as a surrogate for species richness: II. Local applications. *Proceedings of the Royal Society, London (B)* 263, 1571-1575.

Balmford, A., Mace., G.M. & Ginsberg, J.R. (1998). The challenges to conservation in a changing world: putting processes on the map. In *Conservation in a changing world.* (Eds. G.M. Mace, A. Balmford and J.R. Ginsberg), pp 1-28. Cambridge University Press, UK.

Baudry, J. (1993). Landscape dynamics and farming systems: Problems of relating patterns and predicting ecological changes. In *Landscape Ecology & Agroecosystems*. (Eds R.G.H. Bunce, L. Ryszkowski & M.G. Paoletti), pp 21-40. Lewis Publishers, London.

Bedward, M., Pressey, R.L. & Keith, D.A. (1992). A new approach for selecting fully representative reserve networks: addressing efficiency, reserve design and land suitability with an iterative analysis. *Biological Conservation* 62, 115-125.

Belbin, L. (1993). Environmental representativeness: regional partitioning and reserve selection. *Biological Conservation* 66, 223-230.

Brown, J.H. (1995). *Macroecology*. University of Chicago Press, Chicago.

Butt, M.J., Everard, D.A. & Geldenhuys, C.J. (1994). *The distribution and composition of vegetation types in the Soutpansberg-Blouberg Mountain Complex.* Forestek, CSIR, Pretoria. Report FOR-DEA-814.

Caro, T.M. & O'Doherty, G. (1999). On the use of surrogate species in conservation biology. *Conservation Biology* 13, 805-814.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. & Diaz, S. (2000). Review article: Consequences of changing biodiversity. *Nature* 405, 234-242.

Church, R.L., Stoms, D.M. & Davis, F.W. (1996). Reserve selection as a maximal covering location problem. *Biological Conservation* 76, 105-112.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G. & Ellis, A.G. (1999). From representation to persistence: requirements for a sustainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa. *Diversity & Distributions* 5, 51-71.

Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L., Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Manuela, H. & Sahr, K. (1997). A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80, 83-97.

Dale, V.H., Pearson, S.M., Offerman, H.L. & O'Neill, R.V. (1994). Relating patterns of land-use change to faunal biodiversity in central Amazon. *Conservation Biology* 8, 1027-1036.

DEAT. (1996). *National Protected Areas Database*. Department Environmental Affairs and Tourism, Pretoria.

Di Benedetto, L., Luciani, F., Maugeri, G., Poli Marchese, E. & Razzara, S. (1993). Role of natural vegetation in the agricultural Landscape for biological conservation in Sicily. In *Landscape Ecology & Agroecosystems*. (Eds. R.G.H. Bunce, L. Ryszkowski & M.G. Paoletti), pp 131-138. Lewis Publishers, London.

ESRI. (1998). *Arc and Grid Command References*. Environmental Systems Research Institute, Redlands, California.

Fairbanks, D.H.K. (1997). Regional *Eucalyptus* species suitability mapping: Western Cape and hybrids for summer rainfall zones. Environmentek, CSIR, Pretoria. DWAF Report ENV/P/C 97271.

Fairbanks, D.H.K. & Benn, G.A. (2000). Identifying regional landscapes for conservation planning: A case study from KwaZulu-Natal, South Africa. *Landscape & Urban Planning* 50, 1-19.

Fairbanks, D.H.K. & Smith, R.E. (1995). *Report and maps on the afforestation potential land evaluation on the Northern Province, Western Cape Province, Ciskei and Transkei*. Forestek, CSIR, Pretoria. Report FOR-DEA 940.

Fairbanks, D.H.K. & Thompson, M.W. (1996). Assessing land-cover map accuracy for the South African land-cover database. *South African Journal of Science* 92, 465-470.

Fairbanks, D.H.K., Thompson, M.W., Vink, D.E., Newby, T.S., van den Berg, H.M. & Everard, D.A. (2000). The South African land-cover characteristics database: a synopsis of the landscape. *South African Journal of Science* 96, 69-82.

Faith, D.P. (1992). Conservation evaluation and phylogenetic diversity. *Biological Conservation* 61, 1-10.

Faith, D.P. (1994). Phylogenetic diversity: a general framework for the prediction of feature diversity. In *Systematics & Conservation Evaluation* (Eds. P.L. Forey, C.J. Humphries & R.I. Vane-Wright), pp 251-268. Clarendon Press, Oxford.

Faith, D.P. (1995). *Biodiversity and regional sustainability analysis*. CSIRO, Division of Wildlife and Ecology, Canberra, Australia.

Faith, D.P. & Walker, P.A. (1996a) How do indicator groups provide information about the relative biodiversity of different sets of areas?: on hotspots, complementarity and pattern-based approaches. *Biodiversity Letters* 3, 18-25.

Faith, D.P. & Walker, P.A. (1996b). Environmental diversity: on the best possible use of surrogate data for assessing the relative biodiversity of sets of areas. *Biodiversity & Conservation* 5, 399-415.

Faith, D.P. & Walker, P.A. (1996c). Integrating conservation and development: incorporating vulnerability into biodiversity-assessment of areas. *Biodiversity & Conservation* 5, 417-429.

Faith, D.P. & Walker, P.A. (1996d). Integrating conservation and development: effective trade-offs between biodiversity and cost in the selection of protected areas. *Biodiversity & Conservation* 5, 431-446.

Ferrier, S. Pressey, R.L. & Barrett, T.W. (2000). A new predictor of the irreplaceability of areas for achieving a conservation goal, its application in real-world planning, and a research agenda for further refinement. *Biological Conservation* 93, 303-325.

Flather, C.H., Wilson, K.R., Dean, D.J. & McComb, W.C. (1997) Identifying gaps in conservation networks: of indicators and uncertainty in geographic-based analyses. *Ecological Applications* 7, 531-542.

Forman, R.T.T. (2000). Estimate of the area affected ecologically by the road system in the United States. *Conservation Biology* 14, 31-35.

Forman, R.T.T. & Alexandra, L.E. (1998). Roads and their major ecological effects. *Annual Reviews of Ecology & Systematics* 29, 207-231.

Freemark, K. (1995). Assessing effects of agriculture on terrestrial wildlife: developing a hierarchical approach for the US EPA. *Landscape & Urban Planning* 31, 99-115.

Freitag, S., Hobson, C., Biggs, H.C., & van Jaarsveld, A.S. (1998). Testing for potential survey bias: the effect of roads, urban areas and nature reserves on a Southern African mammal data set. *Animal Conservation* 1, 119-127.

Freitag, S. & Mansell M.W. (1997). The distribution and protection status of selected antlion species (Neuroptera: Mymeleontidae) in South Africa. *African Entomology* 5, 205-16.

Freitag, S., Nicholls, A.O. & van Jaarsveld, A.S. (1996). Nature reserve selection in the Transvaal, South Africa: what data should we be using? *Biodiversity & Conservation* 5, 685-698.

Freitag, S. & van Jaarsveld, A.S. (1995). Towards conserving regional mammalian species diversity: a case study and data critique. *South African Journal of Zoology* 30, 136-144.

Freitag, S., van Jaarsveld, A.S. & Biggs, H.C. (1997). Ranking priority areas: an iterative conservation value based approach. *Biological Conservation* 82, 263-272.

Gaston, K.J. (1996a). Species richness: measure and measurement. In *Biodiversity: a biology of numbers and difference*. (Ed. K.J. Gaston), pp. 77-113. Blackwell Science, Oxford.

Gaston, K.J. (1996b).Biodiversity – congruence. *Progress in Physical Geography* 20, 105-112.

Gaston, K.J. & Spicer, J.I. (1998). *Biodiversity: An introduction*. Chapter 1. Blackwell Science Ltd, Oxford, UK.

Gaston, K.J. & Williams, P.H. (1993). Mapping the world's species - the higher taxon approach. *Biodiversity Letters* 1, 2-8.

Harrison, J.A. (1992). The South African Bird Atlas Project: Five years of growth. *South African Journal of Science* 88, 410-413.

Harrison, J.A., Allan, D.G., Underhill, L.G., Herremans, M., Tree, A.J., Parker, V. & Brown, C.J. (1997). The Atlas of Southern African Birds, Vols. 1 & 2. BirdLife South Africa, Johannesburg.

Heywood, V.H. (1994). The measurement of biodiversity and the politics of implementation. In *Systematics and Conservation Evaluation* (Eds P.L. Forey, C.J. Humphries & R.I. Vane-Wright), Ch 2. Clarendon Press, Oxford, UK.

Hobbs, R. (1993). Effects of landscape fragmentation on ecosystem processes in the Western Australian wheatbelt. *Biological Conservation* 64, 193-201.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W., Baltzer, M., Dickinson, C.J., Lwanga, J.S., Matthews, R.A. & Balmford, A. (1998). Complementarity and the use of indicator groups for reserve selection in Uganda. *Nature* 394, 472-475.

Hull, H.E., Chown, S.L. & Bellamy, C.L. (1998). Identification and evaluation of priority conservation areas for Buprestidae (Coleoptera) in South Africa, Lesotho, Swaziland and Namibia. *African Entomology* 6, 265-274.

Kirkpatrick, J.B. (1983). An iterative method for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation* 25, 127-134.

Koch, S.O., Chown, S.L., Davis, A.L.V., Endrödy-Younga, S. & van Jaarsveld, A.S. (2000). Conservation strategies for poorly surveyed taxa: a dung beetle (Coleoptera, Scarabaeidae) case study from southern Africa. *Journal of Insect Conservation* 4, 45-56.

Kunin, W.E. & Lawton, J.H. (1996). Does biodiversity matter? Evaluating the case for conserving species. In *Biodiversity: A biology of numbers and difference.* (Ed. KJ Gaston), ch 11. Blackwell Science Ltd, Oxford, UK.

Lande, R. (1998). Anthropogenic, ecological and genetic factors in extinction. In *Conservation in a changing world.* (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg), ch 2. Cambridge University Press, Cambridge, UK.

Lombard, A.T. (1995a). Introduction to an evaluation of the protection status of South Africa's vertebrates. *South African Journal of Zoology* 30, 71-82.

Lombard, A.T. (1995b). The problems with multi-species conservation: do hotspots, ideal reserves and existing reserves coincide? *South African Journal of Zoology* 30, 145-163.

Lombard, A.T., Cowling, R.M., Pressey, R.L. & Mustart, P.J. (1997). Reserve selection in a species-rich and fragmented landscape on the Augulhas plain, South Africa. *Conservation Biology* 11, 1101-1116.

Low, A.B. & Rebelo, A. (1996). *Vegetation of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs and Tourism, Pretoria.

Mace, G.M., Balmford, A., Buitani, L., Cowlishaw, G., Dobson, A.P., Faith, D.P., Gaston, K.J., Humphries, C.J., Vane-Wright, R.I., Williams, P.H., Lawton, J.H., Margules, C.R., May, R.M., Nicholls, A.O., Possingham, H.P., Rahbek, C. & van Jaarsveld, A.S. (2000). Its time to work together and stop duplication conservation efforts. *Nature* 405, 393.

Maddock, A. & Du Plessis, M.A. (1999). Can species data only be appropriately used to conserve biodiversity. *Biodiversity & Conservation* 8, 603-615.

Margules, C.R., Cresswell, I.D. & Nicholls, A.O. (1994a). A scientific basis for establishing networks of protected areas. In *Systematics & Conservation Evaluation*. (Eds. P.L. Forey, C.J. Humphries & R.I. Vane-Wright), pp 327-350. Clarendon Press, Oxford.

Margules, C.R., Nicholls, A.O. & Pressey, R.L. (1988). Selecting networks of reserves to maximise biological diversity. *Biological Conservation* 43, 63-76.

Margules, C.R., Nicholls, A.O. & Usher, M.B. (1994b). Apparent species turnover, probability of extinction and the selection of nature reserves: a case study of the Ingleborough limestone pavements. *Conservation Biology* 8, 398-409.

Margules, C.R. & Pressey, R.L. (2000). Review article: Systematic conservation planning. *Nature* 405, 243-253.

Margules, C.R. & Redhead, T.D. (1995). *BioRap: guidelines for using the BioRap methodology and tools*. CSIRO, Australia.

McDonald, D.J. (1997). VEGMAP: a collaborative project for a new vegetation map of southern Africa. *South African Journal of Science* 93, 424-426.

Milton, S.J. & Macdonald, I.A.W. (1988). Tree deaths near tar roads in the Northern Transvaal. *South African Journal of Science* 84, 164-165.

Muller, C. (1999). *The distribution and conservation of termites (Isoptera) and butterflies (Lepidoptera) in South Africa*. Unpublished MSc dissertation. University of Pretoria, Pretoria, South Africa.

Nantel, P., Bouchard, A., Brouillet, L. & Hay, S. (1998). Selection of areas for protecting rare plants with integration of land-use conflicts: A case study for the west coast of Newfoundland, Canada. *Biological Conservation* 84, 223-234.

Nicholls, A.O. (1998). Integrating population abundance, dynamics and distribution into broad-scale priority setting. In *Conservation in a changing world*. (G.M. Mace, A. Balmford & J.R. Ginsberg), pp 251-272. Cambridge University Press, UK

Nicholls, A.O. & Margules, C.R. (1993). An upgraded reserve selection algorithm. *Biological Conservation* 64, 165-169.

Noss, R.F. (1990). Indicators for monitoring biodiversity: a hierarchical approach. *Conservation Biology* 4, 355-64.

Noss, R.F. (1996). Ecosystems as conservation targets. *Trends in Ecology & Evolution* 11, 351.

Owen, D.F. (1971). *Tropical butterflies. The ecology and behaviour of butterflies in the tropics with special reference to African species*. Clarendon Press, Oxford.

Pearson, D.L. & Carroll, S.S. (1999). The influence of spatial scale on cross-taxon congruence patterns and prediction accuracy of species richness. *Journal of Biogeography* 26, 1079-1090.

Pimm, S.L. & Raven, P. (2000). Extinction by numbers. *Nature* 403, 843-845.

Polasky, S., Camm, J.D., Solow, A.R., Csuti, B., White, D. & Ding, R. (2000). Choosing reserve networks with incomplete species information. *Biological Conservation* 94, 1-10.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. (1993). Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature* 356, 335-337.

Pressey, R.L. (1994a). Ad hoc reservation: Forward or backward steps in developing representative reserve systems? *Conservation Biology* 8, 662-668.

Pressey, R.L. (1994b). Land classifications are necessary for conservation planning but what do they tell us about fauna. In *Future of the Fauna of Western New South Wales*. (Eds. D. Lunney, S. Hand, P. Reed & D. Butcher), ch 3. The Royal Zoological Society of New South Wales.

Pressey, R.L. (1997). Priority conservation areas: towards an operational definition for regional assessments. In *National parks and protected areas: selection, delimitation, and management* (Eds. J.J. Pigram & R.C. Sundell), pp 337-357. Centre for Water Policy Research, University of New Engalnd, Australia.

Pressey, R.L. (1998). Algorithms, politics and timber: an example of the role of science in a public, political negotiation process over new conservation areas in production forest. In *Ecology for everyone: communicating ecology to scientists, the public and the politicians*. (Eds. R.T. Willis & R.J. Hobbs), pp 73-87. Surrey Beatty & Sons, Sydney.

Pressey, R.L., Ferrier, S., Hager, T.C., Woods, C.A., Tully, S.L. & Weinman, K.M. (1996). How well protected are the forests of north-eastern New South Wales? – analyses of forest environments in relation to formal protection measures, land tenure and vulnerability to clearing. *Forest Ecology and Management* 85, 311-333.

Pressey, R.L., Humphries, C.L., Margules, C.R., Vane-Wright, R.I., & Williams, P.H. (1993). Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology & Evolution* 8, 124-128.

Pressey, R.L., Johnson, I.R. & Wilson P.D. (1994). Shades of irreplaceability: towards a measure of the contribution of sites to a reservation goal. *Biodiversity & Conservation* 3, 242-62.

Pressey, R.L. & Logan, V.S. (1994). Level of geographic subdivision and its effects on assessments of reserve coverage: a review of regional studies. *Conservation Biology* 8, 1037-1046.

Pressey, R.L. & Logan V.S. (1998). Size of selection units for future reserves and its influence on actual vs targeted representation of features: a case study in western New South Wales. *Biological Conservation* 85, 305-319.

Pressey, R.L. & Nicholls, A.O. (1989). Efficiency in conservation evaluation: scoring versus iterative approaches. *Biological Conservation* 50, 199-218.

Pressey, R.L., Possingham, H.P. & Day, J.R. (1997). Effectiveness of alternative heuristic algorithms for identifying indicative minimum requirements for conservation reserves. *Biological Conservation* 80, 207-219.

Pressey, R.L. & Tully, S.L. (1994). The cost of ad hoc reservation: A case study in western New South Wales. *Australian Journal of Ecology* 19, 375-384.

Pringle, J. (1982). *The Conservationists and the Killers*. T.V. Bulpin and Books of Africa, Cape Town.

Rautenbach, I.L. (1982). Mammals of the Transvaal. *Ecoplan Mongraph No. 1*. Pretoria University Press, Pretoria.

Reijen, R., Foppen, R., Ter Braak, C. & Thissen, J. (1995). The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to the proximity of main roads. *Journal of Applied Ecology* 32, 187-202

Rodrigues, A.S.L., Gregory, R.D. & Gaston, K.J. (2000). Robustness of reserve selection procedures under temporal species turnover. *Proceedings of the Royal Society, London (B)* 267, 49-55.

Rodrigues, A.S.L., Tratt, R., Wheeler, B.D. & Gaston, K.J. (1999). The performance of existing networks of conservation areas in representing biodiversity. *Proceedings of the Royal Society, London (B)* 266, 1453-1460.

Rutherford, M.C. & Westfall, R.H. (1986). Biomes of southern Africa – an objective categorisation. *Memoirs of the Botanical Survey of South Africa* 54, 1-97.

Ryti, R. (1992). Effects of the focal taxon on the selection of nature reserves. *Ecological Applications* 2, 404-410.

SA Surveyor General. (1993a). *Digital elevation model for South Africa*. Department of Land Affairs, Division of Surveying and Mapping, Cape Town, South Africa.

SA Surveyor General. (1993b). *1:500 000 scale road network*. Department of Land Affairs, Division of Surveying and Mapping, Cape Town, South Africa.

Sala, E.S., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. & Wall, D.H. (2000). Global biodiversity scenarios for the year 2100. *Science* 287, 1770-1774.

Schoeman, J.L., Ellis, F., Turner, D.P., MacVicar, C.N., Hartmann, M.O. & Scotney, D.M. (1986). *An overview of crop production potential and erodibility of land in the RSA*. Map Number GB/B/86/12441. Institute for Soil, Climate and Water, Pretoria.

Schulze, R.E. (1998). South African Atlas of Agrohydrology and Climatology Water Research. Commission report TT82/96, Pretoria, South Africa.

Schwartz, M.W. (1999). Choosing the appropriate scale of reserves for conservation. *Annual Review of Ecology and Systematics* 30, 83-108.

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, H., Caicco, S., D'erchia, F., Edwards, T.C., Ulliman, J. & Wright, R.G. (1993). Gap analysis: A geographic approach to protection of biological diversity. *The Journal of Wildlife Management: Wildlife Monographs* No. 123. 57, 1-41.

Smith, H.J. (1998). *Definition and demarcation of high potential land for rain-fed annual crop production.* Unpublished M Inst. Agrar. dissertation. University of. Pretoria, Pretoria, South Africa.

Soulé, M.S. (1991). Conservation: Tactics for a Constant Crisis. *Science* 253, 744-750.

Stoms, D.M. (2000). GAP management status and regional indicators of threats to biodiversity. *Landscape Ecology* 15, 21-33.

Thompson, M. (1996). A standard land-cover classification scheme for remote sensing applications in South Africa. *South African Journal of Science* 92, 34-42.

Tilman, D. (2000). Review article: Causes, consequences and ethic of biodiversity. *Nature* 406, 208-211.

Trombulak, S.C. & Frissell, C.A. (2000). Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14, 18-30.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998). Biodiversity assessments and conservation strategies. *Science* 279, 2106-2108.

Vane-Wright, R.I., Humphries, C.J. & Williams, P.H. (1991). What to protect? Systematics and the agony of choice. *Biological Conservation* 55, 235-254.

Virolainen, K.M., Virola, T., Suhonen, J., Kuitunen, M., Lammi, A. & Siikamaki, P. (1999). Selecting networks of nature reserves: methods do affect the long-term outcome. *Proceedings of the Royal Society, London (B)* 266, 1141-1146.

Wessels, K.J., Freitag, S. & van Jaarsveld, A.S. (1999). The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological Conservation* 89, 21-38.

Wessels, K.J., Reyers, B., Krüger, M. & van Jaarsveld, A.S. (2000). Incorporating land-cover information into regional biodiversity assessments in South Africa. *Animal Conservation* 3, 67-79.

White, D., Minotti, P.G., Barczak, M.J., Sifneos, J.C., Freemark, K.E., Santelmann, M.V., Steinitz, C.F., Kiester, A.R. & Preston, E.M. (1997). Assessing risk to Biodiversity from future landscape change. *Conservation Biology* 11, 349-360.

Whittaker, R.H. (1972). Evolution and the measurement of species diversity. *Taxon* 21, 213-251.

Whittaker, R.H. (1977). Evolution of species diversity in land communities. *Evolutionary Biology* 10, 1-67.

Wickham, J.D., Wu, J. & Bradford, D.F. (1997). A conceptual framework for selecting and analyzing stressor data to study species richness at large spatial scales. *Environmental Management* 21, 247-257.

Williams, P.H. (1998). Key sites for conservation: area selection methods for biodiversity. In *Conservation in a Changing World*. (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg), pp 211-249. Cambridge University Press, UK.

Williams, P.H. & Gaston, K.J. (1994a). Higher taxon surrogacy and richness. *Proceedings of the Royal Society, London (B)* 256, 67-73.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C. & Pressey, R. (1996). A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. *Conservation Biology* 10, 155-174.

Williams, P.H. & Gaston, K.J. (1994b). Measuring more of biodiversity: can higher taxon richness predict wholesale species richness? *Biological Conservation* 67, 211-217.

Williams, P.H. & Humphries, C.J. (1994). Biodiversity, taxonomic relatedness, and endemism in conservation. In *Systematics & Conservation Evaluation* (Eds. P.L. Forey, C.J. Humphries & R.I. Vane-Wright), pp 269-287. Clarendon Press, Oxford.

Wilson, M.G.C. & Anhaeusser, C.R. (Eds.) (1998). *The mineral resources of South Africa: Handbook*, Council for Geoscience, 16, 740pp.

World Commission on Environment and Development (The Brundtland Commission). (1987). *Our Common Future*. Oxford University Press. Oxford, UK.

# CHAPTER 2

# Complementarity as a biodiversity indicator strategy

# Complementarity as a biodiversity indicator strategy

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## Summary

Richness, rarity, endemism and complementarity of indicator taxon species are often used to select conservation areas, which are then assumed to represent most regional biodiversity. Assessments of the degree to which these indicator conservation areas coincide across different taxa have been conducted on a variety of vertebrate, invertebrate and plant groups at a national scale in Britain, Canada, the USA and South Africa and at a regional scale in Cameroon, Uganda and USA. A low degree of spatial overlap among and within these selected indicator conservation areas has been demonstrated. These results tend to suggest that indicator conservation areas display little congruence across different taxa. However, some of these studies demonstrate that many conservation areas for indicator taxa capture a high proportion of non-target species. Thus it appears that indicator conservation areas might sample overall biodiversity efficiently. These indicator conservation areas may, however, exclude species essential for effective conservation, e.g. rare, endemic or endangered species. The present study investigated the value of indicator taxa as biodiversity surrogates using spatial congruence and representativeness of different indicator priority conservation areas. The conservation status of species excluded by the indicator approaches is also assessed. Indicator priority conservation areas demonstrate high land area requirements in order to fully represent non-target species. These results suggest that efficient priority area selection techniques must reach a compromise between maximising non-target species gains and minimising land-use requirements. Reserve selection procedures using indicator based complementarity appear to be approaches which best satisfy this trade-off.

**Introduction**

Concern over the rapid degradation of the world's biological resources and the implications for global biosphere integrity and human welfare is mounting. There is a widely recognised need to design appropriate policy and management strategies to conserve remaining biodiversity resources. The establishment of protected areas for *in situ* biodiversity conservation is one such management strategy. However, the effectiveness of *in situ* conservation strategies depends on the existence of adequate databases about the distribution of species and other natural features. In addition, the need to minimise the costs associated with land acquisition and foregone opportunities for other land-uses when reaching a conservation goal requires efficient procedures for selecting minimum or near minimum sets of sites that represent these species or features (Kirkpatrick, 1983; Bedward *et al.*, 1992; Nicholls & Margules, 1993; Pressey *et al.*, 1993; Freitag & van Jaarsveld, 1997).

Balmford and Gaston (1999) argue that without high quality biodiversity inventories, representative conservation areas will be larger than necessary, thus increasing demands on already limited conservation resources. However, as a rule neither the time nor the resources required to survey all regional biodiversity are available. Thus the selection of representative minimum-set conservation areas often depends on substitute or surrogate biodiversity data which can be surveyed in a more cost and time efficient manner (Noss, 1990; Vane-Wright *et al.*, 1991; Ryti, 1992; Belbin, 1993; Gaston & Williams, 1993; Pressey, 1994; Williams & Gaston, 1994a,b; Margules & Redhead, 1995; Pressey & Logan, 1994; Faith & Walker, 1996; Gaston, 1996b; Williams, 1998). Species-based surrogacy approaches include using measures of species richness, rarity, endemism or complementarity of one or more groups of indicator taxa that have been well surveyed (Prendergast *et al.*, 1993; Lombard, 1995; Williams *et al.*, 1996; Flather *et al.*, 1997; Howard *et al.*, 1998; van Jaarsveld *et al.*, 1998). These approaches assume that a species rich area, areas rich in endemics or complementary areas for indicator groups will be indicative of similar trends in unsurveyed taxa. Consequently, priority conservation areas identified from survey data of one or two indicator groups are capable of conserving most regional biodiversity.

These assumptions of surrogacy require rigorous testing before their implementation. One route to assessing the value of potential indicator taxa is to quantify the degree to which spatial patterns of species richness, endemism, rarity and complementarity coincide across different taxa (Prendergast *et al.*, 1993; Lombard, 1995, Gaston, 1996a; Flather *et al.*, 1997). Although it seems that the distribution of well-studied taxa can act as indicators for the distribution of poorly studied taxa at global and continental scales (Scott *et al.*, 1987; 1993; Pearson & Cassola, 1992), at finer scales (e.g. national and regional) this assumption appears questionable. Prendergast *et al.* (1993), and Prendergast and Eversham (1997) did not find general support for the use of indicator taxa in their British studies, as species richness hotspots (10 km$^2$ grid cell sets) for various vertebrate and invertebrate taxa did not coincide. Similarly in South Africa, Lombard (1995) demonstrated a lack of congruence of species richness, endemism and rarity hotspots (sets of 26km x 26km grid cells) within and among six vertebrate taxa. Williams *et al.* (1996) found that bird richness hotspots were not efficient at representing all British birds, while Williams and Gaston (1998) using 10 $km^2$ grid cell richness data on British fauna agree that the value of indicator taxa for biodiversity conservation planning is far from established. Van Jaarsveld *et al.* (1998) discovered limited overlap between 26km x 26km grid cells selected in South Africa using species richness, rarity and complementarity measures between various vertebrate, invertebrate and plant taxa. In a qualitative assessment of richness hotspots for the USA and Canada for a variety of vertebrate, invertebrate and plant taxa, Flather *et al.* (1997) found a general lack of overlap between cross taxon hotspots. Lawton *et al.* (1998) found that no single vertebrate or invertebrate taxon served as a good indicator for changes in species richness of other taxa with changing disturbance levels in Cameroon.

These results seem to suggest that at a scale relevant to practical conservation planning, the use of indicator taxa for biodiversity conservation has limited potential. However, although hotspots display little congruence among taxa and are less efficient at representing the full complement of species than complementarity approaches (Kirkpatrick, 1983; Margules *et al.*, 1988; 1994; Pressey & Nicholls, 1989; Bedward *et al.*, 1992; Nicholls & Margules, 1993; Freitag *et al.*, 1997; Pressey *et al.*, 1997; van Jaarsveld *et al.*, 1998), conservation planning in the real world is only able to protect a limited number of sites (Reid, 1998). The question then is what proportion of overall diversity can be captured in these conservation areas identified by hotspot approaches.

The previously mentioned studies appear to undermine the use of indicator groups, however, when viewed from an alternative perspective, priority conservation areas for an indicator taxon appear to sample overall biodiversity quite efficiently. Both Prendergast *et al.* (1993) and Lombard (1995) showed that a high proportion of species was captured within priority areas for other taxa, ranging from 48 to 100% ($\bar{x}$ = 80.4%) and 66 to 92% respectively. In Oregon, USA, complementary areas representing one taxon were good at representing the diversity of other terrestrial taxa (Unpublished data in Csuti *et al.*, 1997). Similarly, Howard *et al.* (1998), using the approach developed by Williams *et al.* (2000), found that despite little spatial congruence in species richness of a variety of taxa in Uganda, complementary areas chosen using information on one taxon effectively captured overall diversity. Thus spatial overlap in areas based on species richness of different taxa may be an inadequate assessment of the value of across taxon biodiversity indicator value (Balmford, 1998; Howard *et al.*, 1998). Possibly measures of degrees of representativeness (how completely the reserve system includes the species pool of a region (Margules & Usher, 1981)) of various taxa within indicator areas is a more appropriate method of assessment. Areas containing high levels of diversity for one indicator taxon selected by richness, rarity or complementarity approaches are likely to include a diversity of habitats and therefore a large amount of diversity for other taxa (Reid, 1998).

One shortcoming of this approach towards assessing the value of indicator taxa is that although indicator derived conservation areas may capture a large amount of regional diversity they may be missing species essential for effective conservation, e.g. rare or endangered species. Consequently, richness hotspots may capture a high percentage of overall species diversity, but many rare species do not occur in these hotspots (Prendergast *et al.*, 1993). Red Data Book listed species and endemic species in South Africa were not well represented within hotpots (Lombard, 1995). The distributions of rare species were found to be not strongly nested within the distributions of more widespread species in a study on British birds (Williams *et al.*, 1996). Endangered species hotspots in the USA rarely captured endangered species of other taxa and at least half of the rare species do not occur in hotspots in Australia and Britain (Curnutt *et al.*, 1994; Dobson *et al.*, 1997).

The present study investigated the across taxon value of indicator taxa using spatial congruence and representativeness of richness hotspots, rarity hotspots as well as areas selected by complementarity based richness and rarity algorithms. In addition, a critical evaluation of the conservation status of species overlooked by indicator conservation areas was conducted.

## Methods

*Study area and databases*

The study area comprises the Northern Province of South Africa (Figure 1), which represents approximately 10% (122305 km$^2$) of one of the most biologically rich nations in the world (WCMC, 1992). Information on species presence within 26km x 26km grid cells (*ca.* 700 km$^2$; n = 215) was collated for birds (Aves), butterflies (Lepidoptera: superfamilies Hesperioidea, Papilionoidea), mammals (Mammalia) and vascular plants (Plantae) (Table 1). These taxa are all well surveyed within the study area and reflect little survey bias (Harrison, 1992; Freitag & van Jaarsveld, 1995; Freitag *et al.*, 1998) with the possible exception of the butterfly dataset which contains the lowest number of records surveyed in the fewest grid cells (Table 1). The butterfly dataset is the best available invertebrate dataset for the study region and has the additional advantage of being a taxonomically well-known group (Muller, 1999). Only endemic plant species (species that were not recorded outside of the former Transvaal Province) were included in the analyses, since the representation of all plant species sets unattainable formal conservation goals, requiring over 50% of the study area to represent all species once. All grid cells have been surveyed for plant species; however, only 88.4% of the grid cells contain endemic plant species (Table 1).

*Priority conservation area identification*

Richness and rarity hotspots were identified within the study area for all four taxa separately, as well as for all taxa combined. Richness hotspots were defined as the 5% richest grid cells containing records for that particular taxon or group of taxa. Rarity hotspots were identified as grid cells containing rare species defined by Gaston (1994) as the 25% species with the lowest abundances or number of distribution records (Williams *et al.*, 1996).

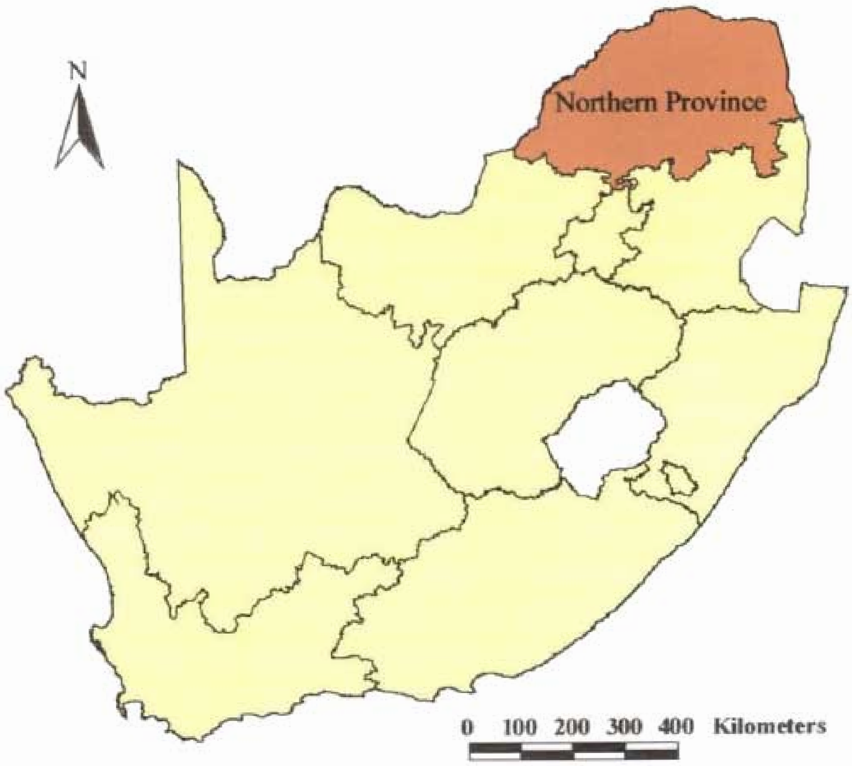


**Figure 1:** Map of South Africa showing location of Northern Province study area.

**Table 1:** Species distribution data.

| Taxon | Unique records | Unique species | Rare species | Endemic species | Grids with records |
|---|---|---|---|---|---|
| Birds | 49089 | 574 | 141 | 63 | 214 (99%) |
| Butterflies | 2062 | 328 | 79 | 4 | 84 (39.1%) |
| Mammals | 5218 | 214 | 56 | 1 | 183 (85.1%) |
| Plants | 2694 | 472 | 125 | 472 | 190 (88.4%) |
| Combined | 59063 | 1588 | 353 | 540 | 215 (100%) |

However, from a conservation perspective it is the overall regional biodiversity that is of interest, not just the extremes of the diversity continuum represented within hotspots (Gaston, 1996a). For this reason the principle of complementarity, which recognises the identity of units or species within grid cells, is included in this study. Complementary sets of grid cells representing all species at least once were identified using a rarity-based complementarity algorithm based on Nicholls and Margules (1993) as well as a richness-based complementarity algorithm. These reserve selection procedures are based on simple heuristic algorithms which proceed in a stepwise fashion, adding grid cells on at each step that contain features most complementary to those in the grid cells already reserved. The algorithms are essentially similar, varying in their point of departure. The former starts with grid cells containing unique features and adds sites progressively according to which contains the rarest unrepresented feature (Nicholls & Margules, 1993). The richness-based algorithm begins with the most species rich grid cell and sequentially includes grid cells that add the most unrepresented species (Kirkpatrick, 1983; Howard *et al.*, 1998). These algorithms were run on all four taxa separately and then on all taxa combined.

*Spatial congruence in species diversity*

The degree of spatial overlap among conservation networks varies substantially, but consistently, when using different measures (Chapter 3). A measure of proportional overlap used by Prendergast *et al.* (1993) and Lombard (1995) provides the most appropriate assessment.

$$\textit{Proportional overlap} = N_c / N_s$$

where: $N_c$ is the number of common grid cells in a pair of priority areas and $N_s$ is the number of grid cells in the smallest priority set of areas containing data for both groups, i.e. the maximum number of overlapping grid cells possible.

As pointed out by Pressey *et al.* (1993), Margules *et al.* (1994) and Williams *et al.* (1996), flexibility is an inherent characteristic of most complementary sets of areas. Thus perhaps measures of proportional overlap are not sufficient in comparing overlap between complementary sets. Few studies have been conducted on the similarities of sets of complementary areas based on different taxa, providing limited evidence of similarities (Ryti, 1992; Saetersdal *et al.*, 1993; Vane-Wright *et al.*, 1994; Gaston *et al.*, 1995). A method similar to that of Williams *et al.* (1994) and Gaston *et al.* (1995), using the selection order of grid cells for complementary sets as an indication of the grid cell's diversity value (in terms of richness or rarity; and complementarity), is applied. The grid cells selected first would thus be assumed to have the highest diversity value. An evaluation of the sequences of grid cells selected for pairs of complementary sets allows for a comparison of patterns of between-taxon diversity. The selection orders of the richness- and rarity-based complementary algorithms were analysed by Pearsons product moment correlations.

*Species representation*

The number of species falling into priority conservation areas was calculated for each of the four taxa as well as for all taxa combined. The number of additional grid cells required to represent all taxa once was calculated. The performance of priority sets in representing overall diversity was evaluated following the approach developed by Williams and colleagues. (Williams *et al.*, 2000), and subsequently employed by Howard *et al.* (1998). The manner in which cumulative percentage species increased as a function of cumulative percentage grid cells selected was determined. This was done for all indicator groups, richness and rarity hotspots, as well as their complementary areas selected using richness- and rarity-based algorithms.

*Rare and endemic species representation*

The ability of the various indicator based priority conservation areas to represent rare and endemic species was investigated. Endemic butterfly, mammal and plant species were defined as species occurring only within the former Transvaal province and rare species as the lowest quartile of species based on distribution records or abundances as in Gaston (1994). There are no birds restricted to the former Transvaal province, thus endemic birds were defined as birds occurring only in South Africa (Table 1). The percentage rare and endemic species represented within the priority conservation areas was calculated. The relationship between cumulative representation of rare and endemic species and the number of grid cells selected within each priority conservation area was examined using an approach similar to that of Williams *et al.*, (2000). The rate at which species and especially rare and endemic species are represented within priority conservation areas could then be ascertained.

## Results

*Priority conservation areas*

Table 2 shows the percentage of grid cells required for priority conservation areas based on all four indicator groups, as well as for all groups combined. The grid cell requirements for these conservation areas vary from 1.9% for the butterfly richness hotspots to 81.9% for the bird rarity hotspots. In general, rarity hotspots required many grid cells while richness hotspots required fewer grid cells. The richness- and rarity-based complementarity networks contained almost identical numbers of grid cells. The birds and combined taxa required the most grid cells within the richness and rarity hotspots while the combined taxa and endemic plants required the most grid cells within the richness and rarity-based complementary networks.

*Spatial congruence in species diversity*

The measure of proportional spatial congruence suggests a high degree of spatial overlap between pairs of priority conservation areas (Figure 2a), and a moderate degree of overlap among priority conservation areas based on indicator taxa (Figure 2b). Overlap between rarity-based complementary networks and rarity hotspots was highest, with overlap between rarity-based complementary networks and richness-hotspots being lowest (Figure 2a). Rarity hotspots and richness hotspots demonstrate the highest and lowest overlap between indicator groups, respectively (Figure 2b). The selection order of the complementary sets of grid cells showed no significant correlations between taxa. The richness- and rarity-based complementary networks based on the same taxa were significantly positively correlated ($r > 0.8$; $p < 0.05$).

*Species representation*

The percentage of species captured in priority conservation areas was high (Table 2); ranging from 59.2% for butterfly richness hotspots to 99.9% for the richness hotspots based on all taxa combined. This excludes the 100% representation achieved by the richness and rarity-based algorithms run on all taxa combined, as these algorithms run until the target representation of 100% of species is achieved (Table 2). Richness hotspots display the lowest degree of species representativeness ($\bar{x}$ = 75.2%) with rarity-based complementary networks, richness-based complementary networks and rarity hotspots displaying higher average species representation percentages across all indicator taxa ($\bar{x}$ = 91.4, 92.1 and 96.3%, respectively).

The additional grid cells required to represent all species at least once range from 0.5% for rarity hotspots based on all taxa combined to 39.1% for butterfly richness hotspots (Table 2). The total percentage grid cells required (i.e. grid cells selected as part of priority conservation areas and additional grid cells required to represent all species once) are similar for the various priority conservation areas (*ca.* 41%), with the exception of the rarity hotspots for all taxa combined and for birds (55.3 and 83.7%, respectively) (Table 2).

Although it would appear that the percentage of species excluded by the priority conservation areas is low (Table 2), upon closer examination these species are primarily rare and endemic species. Out of the species from non-target groups excluded by the indicator priority conservation areas, on average 77.6, 76.5, 92.1 and 90.7% are rare and endemic species missed by richness hotspots, rarity hotspots, richness-based complementary networks and rarity-based complementary networks, respectively (Table 2). From a different perspective, the richness hotspots, richness-based complementary networks, rarity-based complementary networks and rarity hotpots for indicator groups exclude on average 51.4, 21.7, 23.7 and 8% of the rare and endemic species from non-target groups, respectively (Table 2).

**Table 2:** Results on efficiency, representativeness and rare and endemic species representation within the priority conservation areas selected

| Priority conservation areas | % grid cells selected | % total species represented | % additional grid cells to represent all species | % total grids to represent all species | % excluded species that are rare & endemic | % excluded species that are common | % rare & endemic species represented | % rare & endemic species excluded |
|---|---|---|---|---|---|---|---|---|
| **Richness hotspots** | | | | | | | | |
| All taxa | 5.12 | 82.93 | 35.81 | 40.93 | 90.04 | 9.96 | 64.48 | 35.52 |
| Birds | 5.12 | 77.90 | 35.81 | 40.93 | 91.45 | 8.55 | 53.28 | 46.72 |
| Butterflies | 1.86 | 59.19 | 39.07 | 40.93 | 62.96 | 37.04 | 40.61 | 59.39 |
| Mammals | 4.19 | 82.93 | 38.60 | 42.79 | 75.04 | 24.96 | 26.93 | 73.07 |
| Plants | 4.65 | 73.24 | 36.28 | 40.93 | 68.47 | 31.53 | 57.64 | 42.36 |
| ***Average*** | ***4.19*** | ***75.24*** | ***37.12*** | ***41.30*** | ***77.59*** | ***22.41*** | ***48.59*** | ***51.41*** |
| **Rarity hotspots** | | | | | | | | |
| All taxa | 54.88 | 99.94 | 0.47 | 55.35 | 0.00 | 100.00 | 100.00 | 0.00 |
| Birds | 81.86 | 99.69 | 1.86 | 83.72 | 100.00 | 0.00 | 99.27 | 0.73 |
| Butterflies | 16.74 | 92.07 | 25.12 | 41.86 | 93.65 | 6.35 | 82.82 | 17.18 |
| Mammals | 24.65 | 93.83 | 21.86 | 46.51 | 96.94 | 3.06 | 86.17 | 13.83 |
| Plants | 24.65 | 96.03 | 16.28 | 40.93 | 92.06 | 7.94 | 91.56 | 8.44 |
| ***Average*** | ***40.56*** | ***96.31*** | ***13.12*** | ***53.67*** | ***76.53*** | ***23.47*** | ***91.97*** | ***8.03*** |
| **Richness algorithm** | | | | | | | | |
| All taxa | 40.93 | 100.00 | 0.00 | 40.93 | 0.00 | 0.00 | 100.00 | 0.00 |
| Birds | 11.16 | 85.14 | 31.16 | 42.33 | 93.65 | 6.35 | 65.65 | 34.35 |
| Butterflies | 14.42 | 89.99 | 26.98 | 41.40 | 92.21 | 7.79 | 79.33 | 20.67 |
| Mammals | 12.09 | 87.59 | 29.30 | 41.40 | 92.79 | 7.21 | 71.91 | 28.09 |
| Plants | 30.23 | 97.67 | 12.09 | 42.33 | 89.66 | 10.34 | 96.22 | 3.78 |
| ***Average*** | ***21.77*** | ***92.08*** | ***19.91*** | ***41.67*** | ***92.08*** | ***7.92*** | ***82.62*** | ***21.72*** |
| **Rarity algorithm** | | | | | | | | |
| All taxa | 40.93 | 100.00 | 0.00 | 40.93 | 0.00 | 0.00 | 100.00 | 0.00 |
| Birds | 11.16 | 85.08 | 30.70 | 41.86 | 93.31 | 6.69 | 65.50 | 34.50 |
| Butterflies | 14.42 | 90.11 | 26.98 | 41.40 | 92.76 | 7.24 | 79.48 | 20.52 |
| Mammals | 12.09 | 84.38 | 30.23 | 42.33 | 88.77 | 11.23 | 64.34 | 35.66 |
| Plants | 29.77 | 97.42 | 12.56 | 42.33 | 87.88 | 12.12 | 95.78 | 4.22 |
| ***Average*** | ***21.67*** | ***91.40*** | ***20.09*** | ***41.77*** | ***90.68*** | ***9.32*** | ***81.02*** | ***23.73*** |

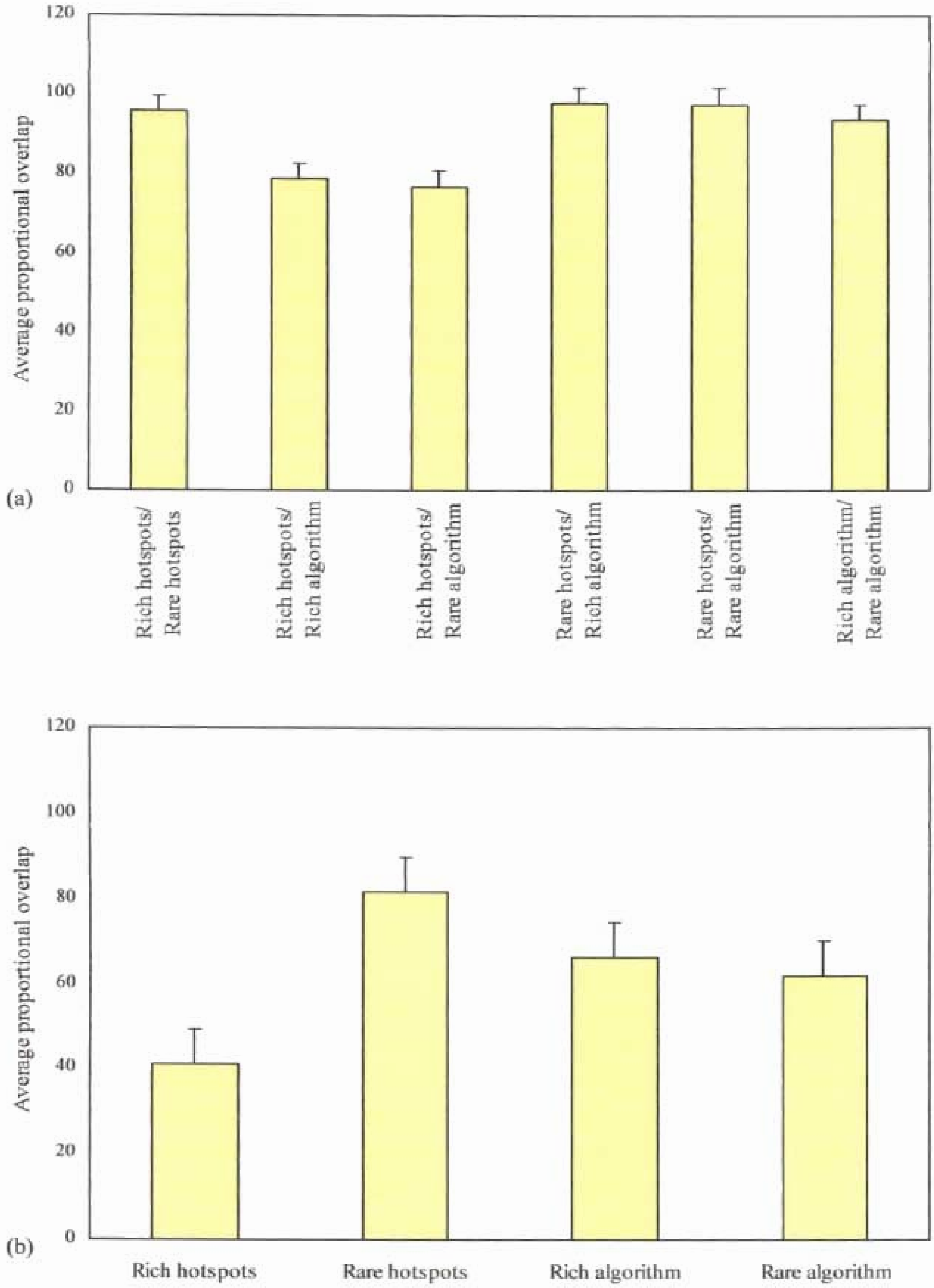


**Figure 2:** The degree of proportional overlap (mean ± s.e; n = 4): (a) between pairs of conservation areas generated by means of different prioritisation criteria (richness and rarity hotspots, richness- and rarity-based complementarity algorithms), and (b) within conservation areas based on different indicator taxa (rich = richness, rare = rarity).

Figure 3 illustrates the rate at which species are represented within the priority conservation areas. The initial rate of representation is rapid, with an average of 70, 87.9, 88 and 86.2% of all species represented within less than 10% of the study area for indicator richness hotspots, rarity hotspots, richness-based complementary networks and rarity-based complementary networks, respectively. The rate then slows dramatically as all priority conservation areas target the representation of all species.

The rate of representation of rare and endemic species is lower than the rate of representation for all species illustrated in Figure 3. This slower rate, with richness hotspots, rarity hotspots, richness-based complementary networks and rarity-based complementary networks respectively capturing 48.6, 71.6, 74.7 and 75.3% of the rare species within 10% of the study area, is demonstrated in Figure 4. The rate also slows further as full representation of all rare and endemic species is targeted.

## Discussion

The results from the present study provide qualified support for the use of indicator taxa in the selection of representative conservation areas. The high levels of spatial congruence are encouraging, but due to the lack of general support from previous studies (Prendergast *et al.*, 1993; Lombard, 1995; van Jaarsveld *et al.*, 1998), this result should be interpreted with caution. The high levels of species representation within the indicator priority conservation areas would appear to support Prendergast *et al.* (1993), Balmford (1998), Howard *et al.*, (1998) and Reid, (1998) in their suggestion that conservation areas species rich for one indicator taxon may represent considerable diversity in other non-target taxa. However, within the species representation analyses as well as within the spatial congruence assessments, the effect of conservation area size is often overlooked. An extensive indicator conservation area has a much higher probability of coinciding with another indicator conservation area, and also stands a greater chance of capturing higher levels of regional biodiversity than restricted conservation areas. This is obvious from the results where complementary networks and rarity hotspots (all large areas) coincide more with one another than with the smaller richness hotspots and also have higher species representation values, capturing more regional species diversity than smaller richness hotspots.

In accordance with findings by Lombard (1995) and Williams *et al.* (1996) richness hotspots contain the highest number of species records per grid cell and thus would appear to be the most effective at representing large numbers of species within fewer grid cells. Taking the present limited state of financial and land resources for conservation into account, this is perhaps an important result. However this result is misleading and should be interpreted with caution. Although richness hotspots may appear to be the most efficient at representing near-maximum regional biodiversity in a minimum number of areas, these richness hotspots exclude up to one-quarter of the species in non-target groups and perhaps more importantly they exclude half the rare and endemic species in non-target groups (Table 2; Figure 5).

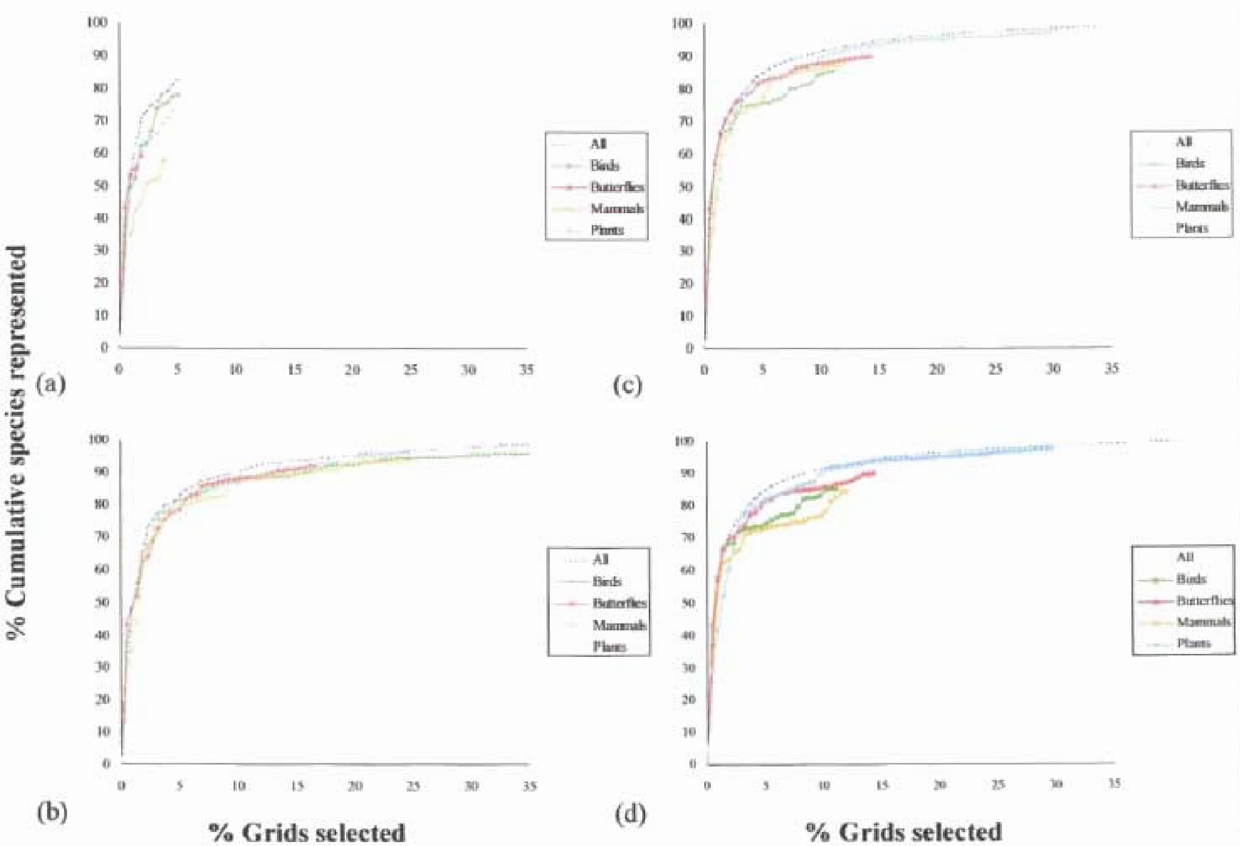


**Figure 3:** Cumulative percentage representation of all four taxa (birds, butterflies, mammals and plants) and all taxa combined as a function of cumulative percentage grid cells selected by (a) richness hotspots, (b) rarity hotspots, (c) the richness-based complementarity algorithm, and (d) the rarity-based complementarity algorithm.

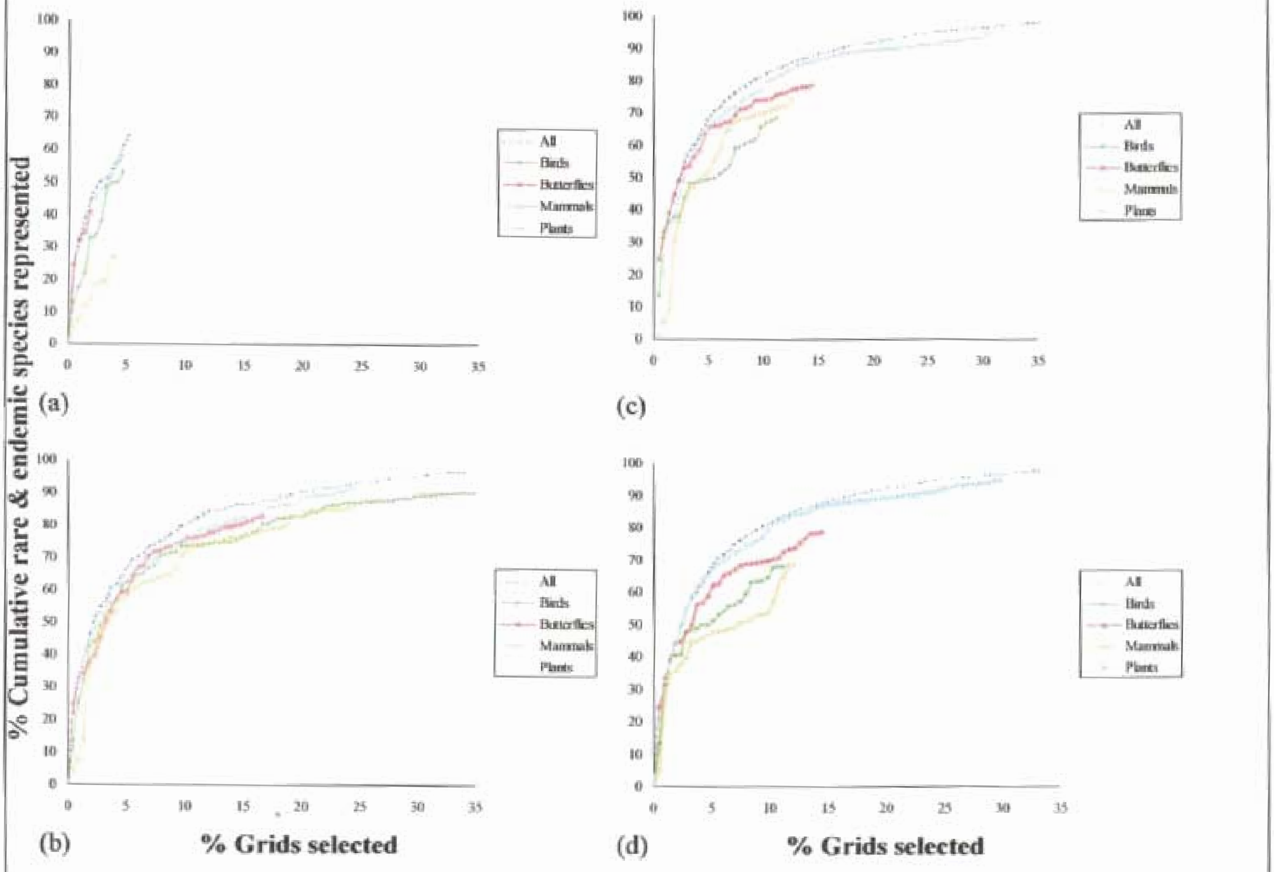


**Figure 4:** Cumulative percentage representation of rare and endemic species of birds, butterflies, mammals, plants and all taxa combined as a function of cumulative percentage grid cells selected by (a) richness hotspots, (b) rarity hotspots, (c) the richness-based complementarity algorithm, and (d) the rarity-based complementarity algorithm.

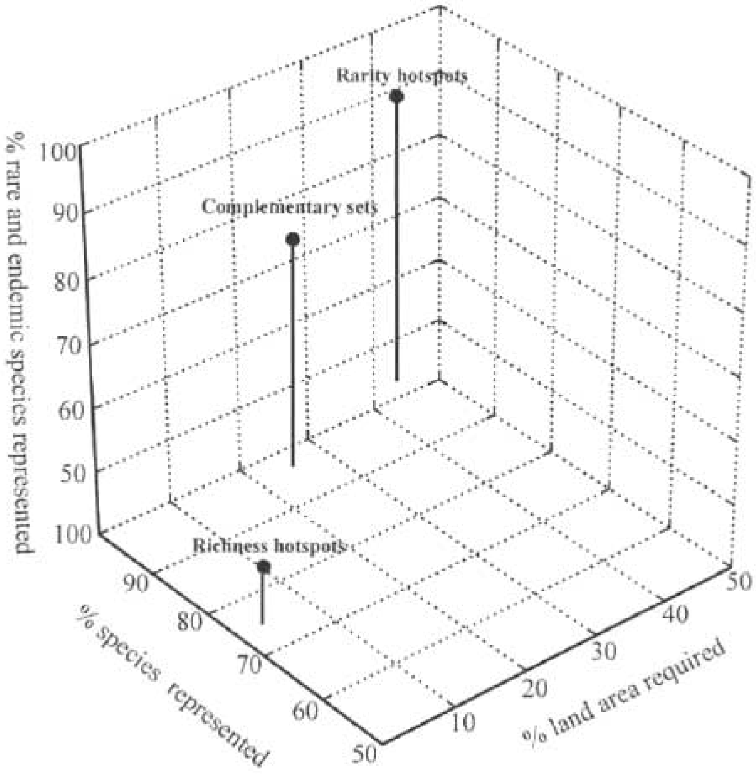


**Figure 5:** Three-dimensional scatterplot of the land-use efficiency and non-target species representation (including rare and endemic species) of richness hotspots, rarity hotspots and complementarity-based reserve selection algorithms.

Rarity hotspots represent species, as well as rare and endemic species, of non-target taxa very well, but this comes at a high land cost, requiring over 40% of the land available (Table 2; Figure 5). Thus it would appear that, as Pressey *et al.* (1993) and Williams (1998) argue, indicator complementary sets of grid cells are perhaps the most efficient conservation solution. These areas protect high levels of non-target biodiversity (92%), missing only 20% of the rare and endemic species (a result similar to the very high levels attained by the land area costly rarity hotspots), in only half the area required by the rarity hotspots (Figure 5).

Although these priority indicator areas appear to efficiently represent a large percentage of regional biodiversity and thus perhaps support the notion of indicator taxa as valuable biodiversity surrogates, two important issues emanating from the present study remain problematic. First, attempts to achieve full representation of all known regional biodiversity will be expensive in terms of land requirements irrespective of which indicator approach is used. This is emphasised by the high number of grid cells (40% of the study region) required to achieve 100% representation of all taxa within all the generated conservation areas. Also, representative networks can be very fragmented and scattered, as is the case with most of the current conservation areas and these highly fragmented or diffuse networks require intensive management and therefore demand high management costs (Bedward *et al.*, 1992).

Second, although species missed by the indicator conservation areas represent a small fraction of the species known to occur within the region, this small component is important in conservation terms. More than half of these excluded species are rare and endemic, and add to the fact that a significant portion of all the rare and endemic species within the region are missed by the various indicator priority conservation networks. Thus, existing methods used to identify indicator priority conservation areas do not seem to be efficient at representing rare and endemic species across taxa and represent them at a very slow rate. This obviously has significant implications for regional conservation planning, as it suggests that the rare and endemic taxa from different groups may be found in different areas (Dobson *et al.*, 1997). It also highlights the need to clarify conservation goals and to decide whether the goal of total species representation, or rare and endemic species representation is the most appropriate one.

**Conclusion**

This study supports the use and importance of indicator taxa as surrogates for regional biodiversity. The occasional lack of cross-taxon congruence between indicator conservation areas (overlap values generally being higher than 90% with values of 76 and 78% between richness-based complementary areas; and rarity and richness hotspots, respectively (Figure 2)), is not sufficient to invalidate the use of indicators as surrogates. High levels of cross-taxon species representativeness within the indicator conservation areas (75-96%) seem to lend support to the assumption that areas of conservation importance to one taxon will capture high levels of diversity for non-target taxa. Although encouraging, this result does not extend to regionally rare and endemic taxa (indicator areas excluding between 8 and

50% of rare and endemic species) and should therefore be implemented with caution. The exclusion of rare and endemic species highlights the need for some form of species specific conservation management. The lack of unqualified support for the indicator taxon strategy, the absence of complete biodiversity inventories and the lack of standard assessment techniques for indicator taxa as surrogates (Flather *et al.*, 1997) all raise important questions about the validity of the surrogate indicator approach.

High levels of species representation, especially of rare and endemic species, appear to come at a cost, requiring large areas of land ranging from 40 to 50% of the land available. This trade-off between land-use efficiency and the representation of species, especially rare and endemic species, suggests that an indicator strategy that manages to reach a compromise between land-use requirements and species representation may be appropriate. It would seem from these assessments that the complementarity indicator approach is still the most efficient approach for maximising non-target species gains in the minimum area possible.

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## References


Balmford, A. (1998). On hotspots and the use of indicators for reserve selection. *Trends in Ecology and Evolution* 13, 409.

Balmford, A. & Gaston, K.J. (1999). Why biodiversity surveys are good value. *Nature* 398, 204-205.

Bedward, M., Pressey, R.L. & Keith, D.A. (1992). A new approach for selecting fully representative reserve networks: addressing efficiency, reserve design and land suitability with an iterative analysis. *Biological Conservation* 62, 115-125.

Belbin, L. (1993). Environmental representativeness: regional partitioning and reserve selection. *Biological Conservation* 66, 223-230.

Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L., Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Huso, M. & Sahr, K. (1997). A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80, 83-97.

Curnutt, J., Lockwood, J., Luh, H.K., Nott, P. & Russell, G. (1994). Hotspots and species diversity. *Nature* 367, 326-327.

Dobson, A.P., Rodriguez, J.P., Roberts, W.M. & Wilcove, D.S. (1997). Geographic distribution of endangered species in the United States. *Science* 275, 550-553.

Faith, D.P. & Walker, P.A. (1996). How do indicator groups provide information about the relative biodiversity of different sets of areas?: on hotspots, complementarity and pattern-based approaches. *Biodiversity Letters* 3, 18-25.

Flather, C.H., Wilson, K.R., Dean, D.J. & McComb, W.C. (1997). Identifying gaps in conservation networks: of indicators and uncertainty in geographic-based analyses. *Ecological Applications* 7, 531-542.

Freitag, S., Hobson, C., Biggs, H.C. & van Jaarsveld, A.S. (1998). Testing for potential survey bias: the effect of roads, urban areas and nature reserves on a southern African mammal data set. *Animal Conservation* 1, 119-127.

Freitag, S. & van Jaarsveld, A.S. (1995). Towards conserving regional mammalian diversity: a case study and data critique. *South African Journal of Zoology* 30, 136-144.

Freitag, S. & van Jaarsveld, A.S. (1997). Relative occupancy, endemism, taxonomic distinctiveness and vulnerability: prioritising regional conservation actions. *Biodiversity & Conservation* 6, 211-232.

Freitag, S., van Jaarsveld, A.S. & Bigss, H.C. (1997). Ranking priority areas: an iterative conservation value based approach. *Biological Conservation* 82, 263-272.

Gaston, K.J. (1994). *Rarity*. London: Chapman and Hall.

Gaston, K.J. (1996a). Biodiversity – congruence. *Progress in Physical Geography* 20, 105-112.

Gaston, K.J. (1996b). Species richness: measure and measurement. *In Biodiversity: a biology of numbers and difference*. (Ed. K.J. Gaston), pp. 77-113. Blackwell Science, Oxford.

Gaston, K.J. & Williams, P.H. (1993). Mapping the world's species - the higher taxon approach. *Biodersity Letters* 1, 2-8.

Gaston, K.J., Williams, P.H., Eggleton, P. & Humphries, C.J. (1995). Large scale patterns of biodiversity: spatial variation in family richness. *Proceedings of the Royal Society, London (B)* 260, 149-154.

Harrison, J.A. (1992). The Southern African Bird Atlas Project databank: five years of growth. *South African Journal of Science* 88, 410-413.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W., Baltzer, M., Dickinson, C.J., Lwanga, J.S., Matthews, R.A. & Balmford, A. (1998). Complementarity and the use of indicator groups for reserve selection in Uganda. *Nature* 394, 472-475.

Kirkpatrick, J.B. (1983). An iterative method for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation* 25, 127-134.

Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larsen, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S. & Watt, A.D. (1998). Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. *Nature* 391, 72-75.

Lombard, A.T. (1995). The problems with multi-species conservation: do hotspots, ideal reserves and existing reserves coincide? *South African Journal of Zoology* 30, 145-163.

Margules, C.R., Cresswell, I.D. & Nicholls, A.O. (1994). *A scientific basis for establishing networks of protected areas.* In *Systematics and conservation evaluation* (Eds. P.L. Forey, C.J. Humphries and R.I. Vane-Wright), pp. 327-350. Oxford University Press, Oxford.

Margules, C.R., Nicholls, A.O. & Pressey, R.L. (1988). Selecting networks of reserves to maximise biological diversity. *Biological Conservation* 43, 63-76.

Margules, C. R. & Redhead, T.D. (1995). *BioRap: guidelines for using the BioRap methodology and tools.* Australia: CSIRO.

Margules, C.R. & Usher, M.B. (1981). Criteria used in assessing wildlife conservation potential: a review. *Biological Conservation* 21, 79-109.

Muller, C. (1999). *The distribution and conservation of termites (Isoptera) and butterflies (Lepidoptera) in South Africa.* Unpublished MSc dissertation. University of Pretoria, Pretoria, South Africa.

Nicholls, A.O. & Margules, C.R. (1993). An upgraded reserve selection algorithm. *Biological Conservation* 64, 164-169.

Noss, R.F. (1990). Indicators for measuring biodiversity: a hierarchical approach. *Conservation Biology* 4, 355-364.

Pearson, D.L & Cassola, F. (1992). World-wide species richness patterns of tiger beetles (Coleoptera: Cicindelidae): indicator taxon for biodiversity and conservation strategies. *Conservation Biology* 6, 376-391.

Prendergast, J.R. & Eversham, B.C. (1997). Species richness covariance in higher taxa: empirical tests of the biodiversity indicator concept. *Ecography* 20, 210-216.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. (1993). Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature* 356, 335-337.

Pressey, R.L. (1994). Land classifications are necessary for conservation planning but what do they tell us about fauna? In *Future of the fauna of western New South Wales* (Eds. D. Lunney, S. Hand, P. Reed and D. Butcher), pp 31-44. Royal society of NSW, Mosman.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993). Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology & Evolution* 8, 124-128.

Pressey, R.L. & Logan, V.S. (1994). Level of geographic subdivision and its effects on assessments of reserve coverage: a review of regional studies. *Conservation Biology* 9, 1506-1517.

Pressey, R.L. & Nicholls, A.O. (1989 Efficiency in conservation evaluation: scoring versus iterative approaches. *Biological Conservation* 50, 199-218.

Pressey, R.L., Possingham, H.P. & Day, J.R. (1997). Effectiveness of alternative heuristic algorithms for identifying indicative minimum requirements for conservation reserves. *Biological Conservation* 80, 207-219.

Reid, W.V. (1998). Biodiversity hotspots. *Trends in Ecology & Evolution* 13, 275-280.

Ryti, R. (1992). Effects of the focal taxon on the selection of nature reserves. *Ecological Applications* 2, 404-410.

Saetersdal, M., Line, J.M. & Birks, H.J.B. (1993). How to maximise biological diversity in nature reserve selection: vascular plants and breeding birds in deciduous woodlands, western Norway. *Biological Conservation* 66, 131-138.

Scott, J.M., Csuti, B., Jacobi, J.D. & Estes, J.E. (1987). Species richness: a geographic approach to protecting future biological diversity. *BioScience* 37, 782-788.

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, H., Caicco, F., D'Erchia, F., Edwards Jr, T.C., Ulliman, J. & Wright, R.G. (1993). Gap analysis: a geographic approach to protection of biological diversity. *Wildlife Monographs* 123, 1-41.

Vane-Wright, R.I., Humphries, C.J. & Williams, P.H. (1991). What to protect? Systematics and the agony of choice. *Biological Conservation* 55, 235-254.

Vane-Wright, R.I., Smith, C.R. & Kitching, I.J. (1994). Systematic assessment of taxic diversity by summation. *In Systematics and conservation evaluation*. (Eds. P.L. Forey, C.J. Humphries & R.I. Vane-Wright), pp 309-326. University Press, Oxford.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998). Biodiversity assessments and conservation strategies. *Science* 279, 2106-2108.

WCMC. (1992). *Development of a National Biodiversity Index: a discussion paper prepared by World Conservation Monitoring Centre. Report on the WCMC*, 15 September, 1992.

Williams, P.H. (1998). Key sites for conservation: area selection methods for biodiversity. In *Conservation in a changing world: integrating processes into priorities for action.* (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg), pp. 211-249. Cambridge University Press, Cambridge.

Williams, P.H., Burgess, N. & Rahbek, C. (2000). Assessing large 'flagship' species for representing the diversity of sub-Saharan mammals, using hotspots of total richness, hotspots of endemism, and hotspots of complementary richness. In *Has the panda had its day? Future priorities for the conservation of mammalian biodiversity.* (Eds. A. Entwistle & N. Dunstone), (*In press*). Symposium of the Zoological Society of London, 14-15 November 1997. Cambridge University Press, Cambridge.

Williams, P.H. & Gaston, K.J. (1994a). Higher taxon surrogacy and richness. *Proceedings of the Royal Society, London (B)* 256, 67-73.

Williams, P.H. & Gaston, K.J. (1994b). Measuring more of biodiversity: can higher taxon richness predict wholesale species richness? *Biological Conservation* 67, 211-217.

Williams, P.H. & Gaston, K.J. (1998). Biodiversity indicators: graphical techniques, smoothing and searching for what makes relationships work. *Ecography* 21, 551-560.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C. & Pressey, R. (1996). A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. *Conservation Biology* 10, 155-174.

# CHAPTER 3

## Assessment techniques for biodiversity surrogates

# Assessment techniques for biodiversity surrogates

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**Abstract**

The use and evaluation of indicator taxa as surrogates for unsurveyed species in the identification of sites important to conservation is a widely researched field. However, support for the use of indicator taxa in reserve selection is often varied and conflicting. We consider that these discrepancies in the levels of support for different indicator approaches are often a result of the assessment techniques employed. Our results appear to confirm the assumption that the assessment technique influences the level of support for indicator taxa as biodiversity surrogates. Because the techniques examine different aspects of indicator approaches, we recommend consideration of final goals and standardisation of techniques before the implementation of indicator-based approaches in conservation planning.

The plight of biodiversity as well as the inadequacy of existing protected areas are well established and much cited facts in conservation biology.[1-7] Existing reserve networks were proclaimed primarily on an *ad hoc* basis and usually contain land with either a low potential for economic and political conflict or a high potential for recreation and tourism.[1,3,8-10] There is therefore a widely recognised need for the systematic establishment of protected areas for effective *in situ* species conservation.[11-16] This realisation has led to the development of systematic, explicit procedures for the selection of representative reserve networks.[14,17-19] These procedures rely on a suitable database of geographical areas of land or water, such as sample sites, grid cells or catchments, containing features of species, habitat types, communities or environments.[20] Frequently, as is the case in this study, the database contains grid cells and unique occurrences of species within those grid cells.[1,6,16,21-24]

Sets of grid cells or reserve networks with a high biodiversity or conservation value can then be identified and selected from this database by a variety of reserve selection procedures. These procedures identify various types of reserve networks which can include richness hotspots (grid cells of high species richness),[21] rarity hotspots (grid cells with high numbers of rare species)[6] and complementary sets of grid cells.[14] Complementary sets are grid cells selected by an iterative algorithm, which proceeds in a step wise fashion adding on grid cells that contain species most complementary (so far unrepresented) to species in grid cells already selected. These algorithms may be either richness-based (initially selecting the richest grid cell)[11,23,25] or rarity-based (initially selecting the grid cell with the rarest species)[14] and then proceed to add grid cells in a complementary fashion until a specific representation target is achieved.

However, the species distribution data used in these reserve selection procedures are often of poor quality, inadequately surveyed and of uncertain taxonomy.[21,25-28] Although it is argued that high quality biodiversity inventories for the selection of representative reserve systems will be more cost efficient in the long run,[29] the resources and time required to conduct these regional biodiversity inventories are usually unavailable. Thus the use of substitute or surrogate biodiversity measures in reserve selection procedures is often recommended. These surrogate measures can be surveyed in a more cost and time efficient manner and include broad-scale environmental measures (e.g. climate or vegetation data), higher taxa (e.g. genera or families) or indicator taxa.[26,30-40] Indicator taxa are groups with a sound taxonomy that have been well surveyed in the region (e.g. birds, butterflies). It is then assumed that patterns of species richness, endemism and rarity in these taxa are indicative of similar patterns in unsurveyed taxa within the region.[26,27,41] Thus by selecting sites for the conservation of indicator taxa (e.g. birds), unsurveyed taxa within the region will also be conserved. This indicator-based surrogacy approach has been widely used in several countries to identify sites of conservation importance.[6,21,22,27,42] However, the assumptions of indicator taxa reflecting regional biodiversity patterns require rigorous testing before reserve networks selected from indicator distribution data can be successfully implemented in practice.

Several assessments of the value of surrogate indicator taxa in reserve selection have been carried out with widely differing results.[6,21,24,25,43,44] Three main types of assessments have been applied in these studies. First, several authors have used the degree of spatial overlap between reserve networks based on different indicator taxa as a measure of the success of these indicator-based networks in including sites of conservation importance for other taxa.[21-23,25,44] Two measures of spatial overlap exist in the conservation biology literature:

(a) *Jaccard coefficient* $= N_c / (N_1 + N_2 - N_c) \times 100$

(b) *Proportional overlap* $= N_c / N_s \times 100$

where: $N_c$ is the number of common sites in a pair of reserve networks, $N_1$ and $N_2$ are the number of sites in the pair of reserve networks and $N_s$ is the number of sites in the smallest reserve network containing data for both groups. Thus the Jaccard coefficient measures spatial congruence as a proportion of the total number of sites selected in both reserve networks,[44] while the measure of proportional overlap measures spatial overlap as a proportion of the maximum overlap possible.[21-23,25,43,45]

Because flexibility is an inherent characteristic of many reserve selection techniques,[6,9,20] the use of spatial overlap is perhaps not an acceptable means of assessing similarities between different indicator-based reserve networks. In recognition of this a technique measuring the Pearson's product moment correlation of the order in which sites are selected by reserve selection procedures based on different indicator taxa has been proposed.[23,46] This selection order of sites for a reserve network indicates the sites' importance in terms of their diversity or conservation values. The sites selected first have a high diversity value because they contain more species, more rare species or more complementary species, depending on the specific reserve selection criteria. This second method of assessment of indicator taxon validity compares the selection order of sites within reserve networks, i.e. the diversity value of those sites, across indicator taxa and thus allows for the assessment of site similarities among different indicator taxa.

The third type of assessment applied in the more recent conservation literature measures the species representativeness of each indicator-based reserve network.[6,21-23,25,27,40,43,47-49] This technique investigates how well each reserve network, identified on the basis of one indicator taxon, captures non-target species or overall biodiversity in the region. This technique can also include evaluation of the representation of rare and endemic species in the indicator-based reserve networks and determines how well these networks represent species essential to effective conservation.[6,21-23,50,51]

These different types of analyses have been conducted on various indicator taxa in many regions of the world and all provide differing levels of support for the surrogate value of indicator taxa. A lack of general support, because of low levels of spatial congruence, has been demonstrated in several studies.[21-23,40,43-45] Conversely, high levels of support for indicator taxa as biodiversity surrogates have been demonstrated using measures of species representativeness.[21-23,25] Many reasons have been advanced for the differing levels of support found in the various studies, including the fact that studies conducted in different regions or using different indicator taxa will not agree with one another.[25,47] However we consider that the range of assessment techniques employed to evaluate the validity of surrogate indicator taxa has a large impact on the outcomes of the studies conducted. Thus in the present study we applied all the assessment techniques discussed above to a suite of indicator-based reserve networks, keeping the study region and taxa involved constant, and evaluated the degree of support provided by each technique to indicator taxa as biodiversity surrogates.

Using 59063 unique distribution records of 1588 bird, butterfly, mammal and vascular plant species in 15' x 15' grid cells (*ca.* 700 $km^2$) in the Northern Province of South Africa we identified a variety of indicator-based reserve networks. These networks were identified by selection procedures using distribution data on indicator taxa of birds, butterflies, mammals and vascular plants. The procedures used these indicator taxa, separately and collectively, to identify different types of reserve networks including richness hotspots, rarity hotspots, and complementary sets of grid cells selected by rarity and richness-based complementary reserve selection algorithms. The degree of similarity within each network type based on different indicator taxa was then assessed, e.g. richness hotspots for birds, butterflies, mammals, plants and all taxa combined were assessed for spatial overlap, selection order correlation and species (including rare and endemic species) representation.

Figure 1 illustrates the widely varying levels of support for the indicator-based surrogate approach assessed using different indicator assessment techniques. Similar to findings by van Jaarsveld *et al.*,[44] Jaccard coefficients of overlap are low ($\bar{x}$ = 28%), and although measures of proportional overlap are higher ($\bar{x}$ = 63%), these results indicate the relatively low levels of spatial overlap between reserve networks based on different indicator taxa and are in agreement with the low values of spatial congruence found in previous studies.[21-23,25,44]

Thus the low levels of spatial congruence appear to question the value and perhaps reject the notion of indicator taxa as biodiversity surrogates. The low degree of similarity ($\bar{x}$ = 9%) of selection orders within each reserve network type based on different indicator taxa is similar to that found by Reyers *et al.*[23] (Chapter 2) and Gaston *et al.*,[46] suggesting very different conservation or diversity values for different taxa within each site, providing limited support for indicator taxa as surrogates.

The high levels of species representativeness ($\bar{x}$ = 89%) within the indicator-based reserve networks suggest that a network based on one particular taxon captures high levels of non-target species within the region. This appears to support the use of indicator taxa as surrogates and is in agreement with the high levels found by Prendergast *et al.*[21] and Howard *et al.*[25] Although moderately high numbers of rare and endemic species are represented within the reserve networks ($\bar{x}$ = 76%), of the species excluded by the reserve networks an average of 84% are rare and endemic, which casts doubt on the validity of these networks in the effective conservation of all regional biodiversity.[6,23,50,51]

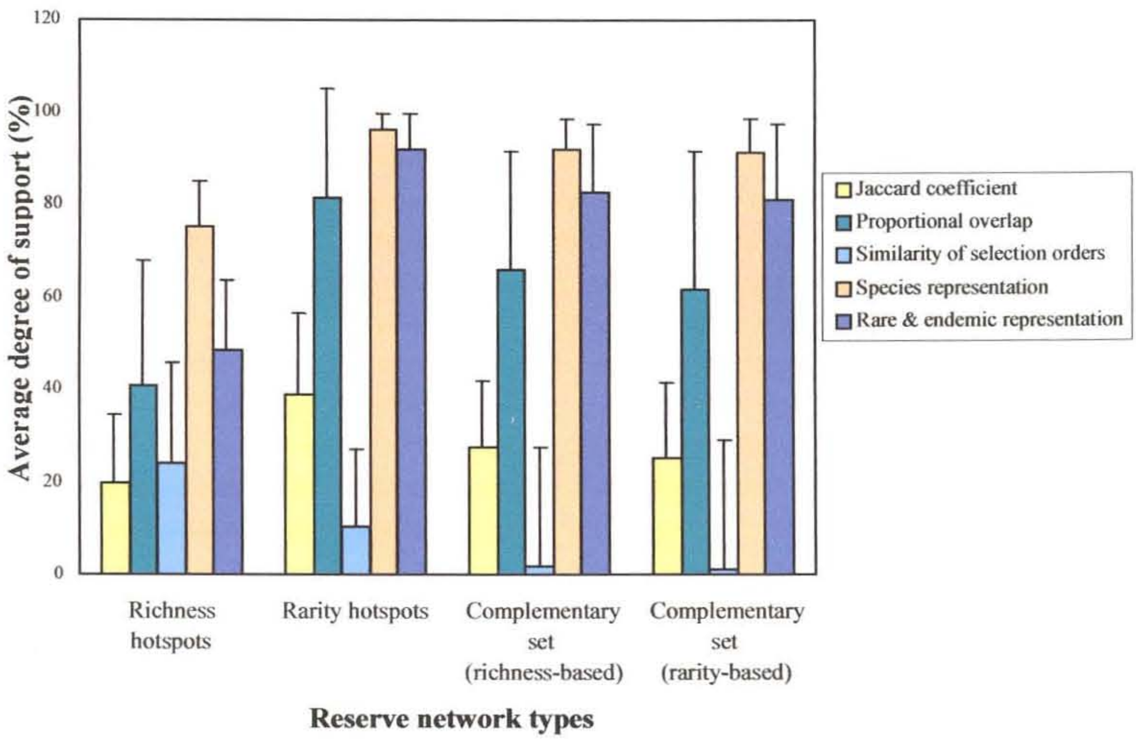


**Figure 1:** Reserve network types (x axis) made up of 15' x 15' grid cells were selected by a suite of reserve selection procedures using distribution data of four indicator taxa, separately and collectively, in the Northern Province of South Africa. Each indicator-based reserve network of each network type was then evaluated in terms of the percentage spatial overlap between networks (measured by Jaccard coefficients and proportional overlap measures), percentage similarity of selection orders of sites within networks and percentage total species (including rare and endemic species) sampled within each network. The percentages were then aggregated and averaged for each network type to illustrate the average degree of support (y axis) provided by the various indicator assessment techniques

It thus appears that the biodiversity indicator assessment technique used influences the strength of support for indicator taxa as biodiversity surrogates. This is an obvious outcome, as each assessment technique is, in fact, testing a different facet of the indicator-based reserve network. Jaccard coefficients and measures of proportional overlap evaluate spatial aspects of indicator-based networks; selection order correlations assess the taxon specific diversity or conservation value of sites; and species representation (including rare and endemic species representation) is an indication of how completely the regional species pool is sampled. All methods have their strong and weak points and although measures of congruence have important implications for conservation, key measures are those that look at species capture (all species but especially rare and endemic species).

The lack of spatial congruence between reserve networks based on complementarity, richness, rarity and endemism, illustrates that indicator taxa cannot be relied on to illustrate similar trends in other, unsurveyed groups. The reason for this could be that reserve networks selected for a particular taxon at a regional and local scale often contain species with narrow habitat requirements (e.g. rare and endemic species), and thus these networks are not likely to correspond across different taxa.[48] This would also explain the different diversity values of each site within reserve networks for a specific taxon demonstrated by the highly dissimilar selection orders of sites for different indicator-based reserve networks. However, if one considers the amount of non-target species recorded within the indicator-based reserve networks, the picture is far less bleak, perhaps because representing a high level of diversity within one taxon, samples a large number of varied habitats and therefore also represents diversity in other unsurveyed taxa.[25,47,48] The high numbers of rare and endemic species excluded support the notion that species from one taxon with narrow habitat requirements do not coincide across taxa and suggest the need for caution before implementation of these principles in conservation planning.

In conclusion it appears that the biodiversity indicator assessment technique used does influence the degree of support for the use of indicator taxa as biodiversity surrogates. This, together with the lack of unqualified support for the indicator taxon strategy, raises important questions about the validity of the indicator approach. Finally, we believe there is a need for the standardisation of assessment techniques employed, otherwise levels of support will continue to fluctuate and consensus on the adequacy of indicator taxa as biodiversity surrogates will remain elusive.

## References

1. Rebelo, A. & Siegfried, W.R. (1992). Where should nature reserves be located in the Cape Floristic Region, South Africa? Models for the spatial configuration of a reserve network aimed at maximising the protection of floral biodiversity. *Conservation Biology* 6, 243-252.
2. Kershaw, M., Williams, P.H. & Mace, G.M. (1994). Conservation of Afrotropical antelopes: consequences and efficiency of using different site selection methods and diversity criteria. *Biodiversity & Conservation* 7, 109-135.
3. Pressey, R.L. & Tully, S.L. (1994). The cost of *ad hoc* conservation: a case study in western New South Wales. *Australian Journal of Ecology* 19, 357-384.
4. Castro Parga, I., Moreno Saiz, J.C., Humphries, C.J. & Williams, P.H. (1996). Strengthening the Natural and National Park system of Iberia to conserve vascular plants. *Botanical Journal of the Linnean Society* 121, 189-206.
5. Pressey, R.L., Possingham, H.P. & Margules, C.R. (1996). Optimality in reserve selection algorithms: when does it matter and how much? *Biological Conservation* 76, 259-267.
6. Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C. & Pressey, R. (1996). A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. *Conservation Biology* 10, 155-174.
7. Freitag, S., Nicholls, A.O. & van Jaarsveld, A.S. (1998). Dealing with established reserve networks and incomplete distribution data sets in conservation planning. *South African Journal of Science* 94, 79-86.
8. Pressey, R.L. (1994). Land classifications are necessary for conservation planning but what do they tell us about fauna? In *Future of the fauna of western New South Wales*. (Eds. D. Lunney, S. Hand, P. Reed & D. Butcher), pp. 31-44. Royal society of NSW, Mosman.
9. Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993). Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology & Evolution* 8, 124-128.
10. Rodrigues, A.S.L., Tratt, R., Wheeler, B.D. & Gaston, K.J (1999). The performance of existing networks of conservation areas in representing biodiversity. *Proceedings of the Royal Society, London (B)* 266, 1453-1460.
11. Kirkpatrick, J.B. (1983). An iterative method for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation* 25, 127-134.
12. Bedward, M., Pressey, R.L. & Keith, D.A. (1992). A new approach for selecting fully representative reserve networks: addressing efficiency, reserve design and land suitability with an iterative analysis. *Biological Conservation* 62, 115-125.

13. Virolainen, K.M., Virola, T., Suhonen, J., Kuitunen, M., Lammi, A. & Siikamaki, P. (1999). Selecting networks of nature reserves: methods do affect the long term outcome. *Proceedings of the Royal Society, London (B)* 266, 1141-1146.
14. Nicholls, A.O. & Margules, C.R. (1993). An upgraded reserve selection algorithm. *Biological Conservation* 64, 164-169.
15. Belbin, L. (1995). A multivariate approach to the selection of biological reserves. *Biodiversity & Conservation* 4, 951-963.
16. Freitag, S. & van Jaarsveld, A.S. (1997). Relative occupancy, endemism, taxonomic distinctiveness and vulnerability: prioritising regional conservation actions. *Biodiversity & Conservation* 6, 211-232.
17. Pressey, R.L. & Nicholls, A.O. (1989). Application of a numerical algorithm to the selection of reserves in semi-arid New South Wales. *Biological Conservation* 50, 264-278
18. Pressey, R.L. & Nicholls, A.O. (1989). Efficiency in conservation evaluation: scoring versus iterative procedures. *Biological Conservation* 50, 199-218.
19. Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L., Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Manuela, H. & Sahr, K. (1997). A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80, 83-97.
20. Margules, C.R., Cresswell, I.D. & Nicholls, A.O. (1994). A scientific basis for establishing networks of protected areas. In *Systematics and conservation evaluation.* (Eds. P.L. Forey, C.J. Humphries & R.I. Vane-Wright), pp. 327-350. Oxford University Press, Oxford.
21. Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. (1993). Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature* 356, 335-337.
22. Lombard, A.T. (1995). The problems with multi-species conservation: do hotspots, ideal reserves and existing reserves coincide? *South African Journal of Zoology* 30, 145-163.
23. Reyers, B., van Jaarsveld, A.S. & Krüger, M. (2000). Complementarity as a biodiversity indicator strategy. *Proceedings of the Royal Society, London (B)* 267, 1-9.
24. Nantel, P., Bouchard, A., Brouillet, L. & Hay, S. 1998. Selection of areas for protecting rare plants with intergration of land-use conflicts: a case study for the west coast of Newfoundland, Canada. *Biological Conservation* 84, 223-234.
25. Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W., Baltzer, M., Dickinson, C.J., Lwanga, J.S., Matthews, R.A. & Balmford, A. (1998). Complementarity and the use of indicator groups for reserve selection in Uganda. *Nature* 394, 472-475.
26. Faith, D.P. & Walker, P.A. (1996). How do indicator groups provide information about the relative biodiversity of different sets of areas?: on hotspots, complementarity and pattern-based approaches. *Biodiversity Letters* 3, 18-25.

27. Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larsen, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S. & Watt, A.D. (1998). Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. *Nature* 391, 72-75.
28. Prendergast, J.R., Quinn, R.M. & Lawton, J.H. (1999). The gaps between theory and practice in selecting nature reserves. *Conservation Biology* 13, 484-492.
29. Balmford, A. & Gaston, K.J. (1999). Why biodiversity surveys are good value. *Nature* 398, 204-205.
30. Noss, R.F. (1990). Indicators for measuring biodiversity: a hierarchical approach. *Conservation Biology* 4, 355-364.
31. Vane-Wright, R.I., Humphries, C.J. & Williams, P.H. (1991). What to protect? Systematics and the agony of choice. *Biological Conservation* 55, 235-254.
32. Ryti, R. (1992). Effects of the focal taxon on the selection of nature reserves. *Ecological Applications* 2, 404-410.
33. Belbin, L. (1993). Environmental representativeness: regional partitioning and reserve selection. *Biological Conservation* 66, 223-230.
34. Gaston, K.J. & Williams, P.H. (1993). Mapping the world's species - the higher taxon approach. *Biodiversity Letters* 1, 2-8.
35. Wessels, K.J., Freitag, S. & van Jaarsveld, A.S. (1999). The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological Conservation* 89, 29-38.
36. Williams, P.H. & Gaston, K.J. (1994). Measuring more of biodiversity: can higher taxon richness predict wholesale species richness? *Biological Conservation* 67, 211-217.
37. Margules, C. R. & Redhead, T.D. (1995). *BioRap: guidelines for using the BioRap methodology and tools.* CSIRO, Australia.
38. Pressey, R.L. & Logan, V.S. (1994). Level of geographic subdivision and its effects on assessments of reserve coverage: a review of regional studies. *Conservation Biology* 9, 1506-1517.
39. Gaston, K.J. (1996). Species richness: measure and measurement. In *Biodiversity: a biology of numbers and difference*, (Ed. K.J. Gaston), pp. 77-113. Blackwell Science, Oxford.
40. Williams, P.H. (1998). Key sites for conservation: area selection methods for biodiversity. In *Conservation in a changing world: integrating processes into priorities for action*, (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg), pp. 211-249. Cambridge University Press, Cambridge.
41. Pearson, D.L. & Cassola, F. (1992). World-wide species richness patterns of tiger beetles (Coleoptera: Cicindelidae): indicator taxon for biodiversity and conservation studies. *Conservation Biology* 6, 376-391.
42. Williams, P.H. & Gaston, K.J. (1998). Biodiversity indicators: graphical techniques, smoothing and searching for what makes relationships work. *Ecography* 21, 551-560.

43. Flather, C.H., Wilson, K.R., Dean, D.J. & McComb, W.C. (1997). Identifying gaps in conservation networks: of indicators and uncertainty in geographic-based analyses. *Ecological Applications* 7, 531-542.

44. van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998). Biodiversity assessments and conservation strategies. *Science* 279, 2106-2108.

45. Prendergast, J.R. & Eversham, B.C. (1997). Species richness covariance in higher taxa: empirical tests of the biodiversity indicator concept. *Ecography* 20, 210-216.

46. Gaston, K.J., Williams, P.H., Eggleton, P. & Humphries, C.J. (1995). Large scale patterns of biodiversity: spatial variation in family richness. *Proceedings of the Royal Society, London (B)* 260, 149-154.

47. Balmford, A. (1998). On hotspots and the use of indicators for reserve selection. *Trends in Ecology & Evolution* 13, 409.

48. Reid, W.V. (1998). Biodiversity hotspots. *Trends in Ecology & Evolution* 13, 275-280.

49. Williams, P.H., Burgess, N. & Rahbek, C. (2000) Assessing large 'flagship' species for representing the diversity of sub-Saharan mammals, using hotspots of total richness, hotspots of endemism, and hotspots of complementary richness. In *Has the panda had its day? Future priorities for the conservation of mammalian biodiversity*, (Eds. A. Entwistle & N. Dunstone), (*In press*). Symposium of the Zoological Society of London, 14-15 November 1997. Cambridge University Press, Cambridge.

50. Curnutt, J., Lockwood, J., Luh, H.K., Nott, P. & Russell, G. (1994). Hotspots and species diversity. *Nature* 367, 326-327.

51. Dobson, A.P., Rodriguez, J.P., Roberts, W.M. & Wilcove, D.S. (1997). Geographic distribution of endangered species in the United States. *Science* 275, 550-553.

# CHAPTER 4

## An assessment of biodiversity surrogacy options in the Northern Province of South Africa

# An assessment of biodiversity surrogacy options in the Northern Province of South Africa

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**Abstract**

Because of the inadequacy of existing data on the distribution of biodiversity, surrogate measures for regional biodiversity have long been used in conservation area selection. These measures include species and environmental surrogate measures, which include vegetation types and land systems and classes. However, the assumed relationship between these surrogate measures and regional biodiversity has seldom been demonstrated. This study uses both species and environmental surrogates of vegetation and landtypes in selecting important areas for biodiversity conservation in the Northern Province of South Africa. The success of these measures in capturing known regional biodiversity is evaluated, as well as their success at identifying important areas containing threatened, rare and endemic non-target biodiversity features. The spatial congruence of the areas identified using different surrogate measures was also assessed. A combined approach to reserve selection using both species and environmental measures was also applied and the areas identified were evaluated in terms of their efficient representation of regional biodiversity. There is a trade-off between success at representing non-target biodiversity features (especially rare, threatened and endemic features) and land-use efficiency. Combined approaches have similar levels of success in representing regional biodiversity, although landtype based ones are more land-use inefficient. The trade-off between efficiency and representation suggests that many of the important conservation areas identified will rely on off-reserve management rather than formal protection. Furthermore, results suggest that recommended national conservation targets of 10% are inadequate

## Introduction

The need to conserve the world's remaining biodiversity is widely recognised, as human impacts threaten an extinction event likely to rival previous mass extinctions of the geological past (Wilson, 1988; UNEP, 1995; Pimm & Raven, 2000). The resources available for the conservation of biodiversity are limited and there is therefore a need to identify priority areas for conservation swiftly and cost-effectively. Several systematic approaches to the identification of conservation areas in which all known regional biodiversity is protected have been suggested (Kirkpatrick, 1983; Margules *et al.*, 1988; Pressey & Nicholls, 1989a; Bedward *et al.*, 1992; Nicholls & Margules, 1993; Margules *et al.*, 1994; Underhill, 1994; Freitag *et al.*, 1996; Church *et al.*, 1996; Csuti *et al.*, 1997; Margules & Pressey, 2000). These approaches, however, require extensive information on the distribution and taxonomy of species. Often regions under evaluation have inadequate databases on species distribution due to poor quality biological survey data and inconsistent basic taxonomy (Haila & Margules, 1996). Conservation area selection techniques must then rely on substitute or "surrogate" measures of biodiversity (Belbin, 1993; Prendergast *et al.*, 1993; Pressey, 1994; Margules & Redhead, 1995; Pressey & Logan, 1995; Balmford, 1998; Howard *et al.*, 1998; Reid, 1998; Pressey *et al.*, 2000).

These surrogate measures include the species surrogate measures of richness, endemism, rarity and complementarity of indicator groups (Margules *et al.*, 1988; Rebelo & Siegfried, 1992; Nicholls & Margules, 1993; Prendergast *et al.*, 1993; Pressey *et al.*, 1993; Margules *et al.*, 1994; Lombard, 1995; Williams *et al.*, 1996; Flather *et al.*, 1997; Howard *et al.*, 1998; Lawton, 1998; van Jaarsveld *et al.*, 1998), higher taxa such as genera or families (Gaston & Williams, 1993; Williams & Gaston, 1994; Williams *et al.*, 1994), or the environmental surrogate measures of vegetation types, land systems or classes and environmental domains (Noss, 1987; Purdie *et al.*, 1986; Belbin, 1993; Pressey, 1994; Pressey & Logan, 1994; Margules & Redhead, 1995; Pressey & Logan, 1995; Faith & Walker, 1996; Wessels *et al.*, 1999; Fairbanks & Benn, 2000; Pressey *et al.*, 2000; Cowling & Heijnis, (In Press)). These environmental surrogate classes are derived from information on vegetation types, soil properties, remote sensing data, climatic data and terrain data (Austin & Margules, 1986; Margules & Redhead, 1995).

Although the need for surrogate measures is widely recognised, there is still no consensus as to which measures are the most applicable. It has also been argued that this assumed relationship between surrogate measures and regional biodiversity be demonstrated before it is put into practice in conservation planning (Pressey, 1994; Williams & Humphries, 1996; Wessels *et al.*, 1999). The aforementioned species surrogate measures have been widely used (Prendergast *et al.*, 1993; Launer & Murphy, 1994; Williams *et al.*, 1996; Freitag *et al.*, 1997; Kerr, 1997), but have several shortcomings including: a lack of correlation between these surrogate measures and regional biodiversity (Chapters 2 and 3) (Rebelo & Siegfried, 1992; Prendergast *et al.*, 1993; Gaston & Williams, 1993; Margules *et al.*, 1994; Williams & Gaston, 1994; Margules & Redhead, 1995; Faith & Walker, 1996; Gaston, 1996) as well as a lack of coincidence between priority conservation areas selected using various species surrogate measures involving different taxa (Chapters 2 and 3) (Lombard *et al.*, 1995; Prendergast *et al.*, 1993; Howard *et al.*, 1998; van Jaarsveld *et al.*, 1998; Mace *et al.*, 2000). This finding suggests that complementary conservation networks selected to represent specific taxa are unlikely to be representative of all biodiversity.

When considering the usefulness of environmental surrogates, Faith and Walker (1996) argue that, if correctly measured, environmental variation should indicate organismal diversity. Thus different environmental classes are assumed to contain different species assemblages. Therefore the protection of these classes should ensure that all or most species within the region will also be protected (Belbin, 1993). The representativeness of conservation area networks has been assessed using various environmental attributes (Scott *et al.*, 1987; Faith & Norris, 1989; Pressey & Nicholls, 1989b; Belbin, 1993; Margules *et al.*, 1994; Pressey & Logan, 1995). However, as Pressey (1994) points out, the assumed relationship between environmental classes and species distribution and abundance is unclear and seldom investigated. In addition, certain species, especially rare species confined to small patches of habitat which are not recognised as distinct environmental classes, may "fall through the coarse-filter" when using broad-scale environmental classes (Noss, 1983; Bedward *et al.*, 1992; Panzer & Schwartz, 1998).

Nevertheless, environmental surrogates have compelling practical advantages, as information on their distribution is cheaper and easier to acquire than detailed species distribution data. Margules and Redhead (1995) also point out that by representing environmental classes some unknown species and known species with unknown distributions may be represented. The shortcomings of species distribution data and the limitations of environmental surrogate measures in the selection of priority conservation areas suggest that perhaps a combination of the two approaches in conservation planning may be advisable.

This study aims to compare the more traditional species based approach to conservation area selection with site selection based on representing specific target levels of environmental surrogate classes within the Northern Province of South Africa. In other words, conservation areas in which all species known to occur regionally are represented at least once will be compared with conservation areas in which specific levels of vegetation and landtypes are represented.

As Pressey and Logan (1995) argue, assessments of conservation area coverage using environmental classes are scale dependent and influenced by the definition of the various environmental classes. Broad scale classes are relatively heterogenous (Scott *et al.*, 1989), thus selection of these coarse classes is still likely to miss much variation. Fine-scale classes are more homogenous and should therefore lead to a better representation of the environmental variation within a network of conservation areas. This study will also examine the influence of mapping scale on the selection of sites based on the coarse environmental classes of vegetation types, as well as the fine scale classes of landtypes.

Finally a set of conservation areas will be selected and evaluated using a combination of species distribution data and environmental surrogates. This combined approach to conservation area selection is similar to that used by Bull *et al.* (1993) in Margules and Redhead (1995) and Lombard *et al.* (1997). The former study identifies a set of sites, referred to as seed points (which include the location of rare or threatened species, existing conservation areas or rest areas for migratory species). Grid cells were then added until a predetermined proportion of each environmental class was contained within the conservation area network. The present study uses known localities of vertebrate, invertebrate and vascular plant species as seed points and then adds grid cells until a predetermined percent of all vegetation types or landtypes are represented within the network. In addition to this, grid cells with specified representations of these vegetation and landtypes will be used as an initial set of sites to which grid cells will be added until all known species within the region occur at least once in the protected areas.

**Methods**

The study area comprises the Northern Province of South Africa (see Figure 1 in Chapter 1).

*Species based approach*

This part of the study incorporates 2060 species with 61329 unique distribution records for invertebrate, vertebrate and vascular plant species. Selection units are quarter degree or 15' x 15' grid cells (n = 215) with an average area of 700 km$^2$ (Table 1).

<u>Species distribution databases</u>

The species data used includes databases on the distribution of taxa that are frequently used as biodiversity indicators, namely mammals, birds, vascular plants and butterflies. These taxa have a relatively sound taxonomy, are well surveyed within the study area and their distribution data are fairly recent (Table 1) (Harrison, 1992; Freitag & van Jaarsveld, 1995; Freitag *et al.*, 1998; Muller, 1999.). Additional data on the distribution of invertebrate species including buprestid beetles, scarab beetles, termites and neuropterans are also included in the analyses, although these taxa are less well known taxonomically, have older distribution records and are less well surveyed within the study area (Table 1) (Freitag & Mansell, 1997; Muller *et al.*, 1997; Hull *et al.*, 1998; Koch, *et al.*, 2000). Because of the large size of the vascular plant dataset (78% of all species) and the disproportionate effect it has on the resultant conservation area networks (43 additional grids cells out of a possible 215 required to protect all plant species), it was decided to exclude all plants not endemic to the study area. The species distribution data used in this chapter were later updated with the removal and addition of some species due to taxonomic changes as well as the discovery of vagrant and exotic species, this explains the slight differences that exist between this database and the ones used in the other chapters.

**Table 1:** A description of the species distribution databases used in the analyses

| Taxon | Species | Records | Grids | Survey date |
|---|---|---|---|---|
| Mammals (Mammalia) | 182 | 4207 | 170 | 1980-1995 |
| Birds (Aves) | 575 | 49427 | 214 | 1980-1992 |
| Vascular plants (Plantae) | 5711 | 42055 | 215 | 1900-1996 |
| Subgroup: Endemic plants | 472 | 2694 | 215 | 1900-1996 |
| Butterflies (Hesperioidea & Papilionoidea) | 328 | 2062 | 84 | 1905-1980 |
| Buprestid beetles (Buprestidae) | 247 | 977 | 119 | 1900-1996 |
| Scarab beetles (Scarabaeinae) | 218 | 1372 | 124 | 1900-1992 |
| Termites (Isoptera) | 16 | 464 | 160 | 1972-1980 |
| Neuropterans (Myrmeleontidae) | 22 | 126 | 41 | 1900-1996 |
| *Combined databases* | | | | |
| All taxa | 7299 | 100690 | 215 | |
| All taxa (excluding non-endemic plants) | 2060 | 61329 | 215 | |

Conservation area selection procedures

A rarity-based conservation area selection algorithm based on that of Nicholls and Margules (1993) was applied. This iterative algorithm begins by selecting grids cells containing unique occurrences of species and proceeds from there in a step wise fashion selecting the grids containing the next most rarest species until all species are represented at least once within the conservation area network. Ties between grid cells are resolved by applying the principles of adjacency of grid cells and complementarity of species content within grid cells respectively.

*Environmental surrogacy approach*

The environmental surrogates of vegetation types and landtypes were used in the present study.

Vegetation types

Low and Rebelo (1996) define a vegetation type as: "a coherent array of communities which share common species (or abundances of species), possess a similar vegetation structure (vertical profile), and share the same set of ecological processes". Vegetation data for the Northern Province were extracted from the national-scale vegetation map of South Africa (Low & Rebelo, 1996). The Northern Province is covered by three biomes (Forest, Grassland and Savanna) and fifteen vegetation types of which Mixed Bushveld, Mopane Bushveld and Sweet Bushveld are the most dominant (see Table 1 in Chapter 1).

Landtypes

Pedosystems are areas with uniform terrain and soil patterns (MacVicar *et al.*, 1974; Land Type Survey Staff, 1986) and are similar to land systems (Christian & Steward 1968; Lawrence *et al.*, 1993), which have been extensively used as environmental surrogates during conservation area evaluation at broad regional scales in Australia (Purdie *et al.*, 1986; Pressey & Nicholls, 1989b; Pressey & Tully, 1994). Climate zones (mapped at 1:250000 scale) have been superimposed upon pedosystem maps to arrive at maps of landtypes covering the majority of South Africa (MacVicar *et al.*, 1974; Land Type Survey Staff, 1986). A landtype therefore delineates an area at 1:250000 scale which displays a marked degree of uniformity with respect to terrain form, soil pattern and climate (MacVicar *et al.*, 1974; Land Type Survey Staff, 1986). Landtype data for the Northern Province were prepared by the Institute for Soil, Climate and Water (ISCW) of the Agricultural Research Council (ARC). A total of 676 different landtypes occur within the study area. Due to the large size of this database as well as the sensitive nature of the data, the landtypes are represented as numeric codes and therefore no specific references can be made to or conclusions drawn about specific landtypes in the present study.

Conservation area selection procedures

The vegetation map and landtypes of the Northern Province were respectively overlaid with the aforementioned 15' x 15' grid cells. The areas of various vegetation and landtypes within each grid cell (n = 215) were subsequently calculated using ArcInfo.

A percentage representation approach similar to that used by Pressey and Nicholls (1989b), and Pressey and Tully (1994) was applied; this approach attempts to sample a nominated percentage area (5-50%) of each vegetation and landtype. The algorithm initially selects the feature covering the smallest total area and thus conforms to the rarity-based algorithm of Margules *et al.* (1988). An over representation constraint rule was designed to restrict overshooting initial target representation levels, as is often the case in conservation area selection procedures (Bedward *et al.*, 1992; Nicholls & Margules, 1993; Wessels *et al.*, 1999). The adjacency constraint rule was also included to resolve ties and ensure larger contiguous areas where possible (Nicholls & Margules, 1993; Lombard *et al.*, 1995; Freitag *et al.*, 1996; Willis *et al.*, 1996).

*Comparison of the species based and surrogacy approaches*

The conservation area network selected by the species based approach was evaluated in terms of the percentage of each vegetation and landtype it contained. Similarly, the percentages of total species captured within the conservation areas selected to represent target levels of the vegetation and landtype classes were also calculated. In addition to this the spatial congruence or overlap of the conservation areas selected by both species and environmental surrogate based approaches was compared using measures of proportional overlap (Chapter 2 and 3) (Prendergast *et al.*, 1993; Lombard *et al.*, 1995)

$$\textit{Proportional overlap} = N_c / N_s \times 100$$

where: $N_c$ is the number of common sites in a pair of conservation area networks and $N_s$ is the number of sites in the smallest network containing data for both groups. Thus the measure of proportional overlap measures spatial overlap as a proportion of the maximum overlap possible (Chapter 2 and 3). Finally the success of the conservation area networks based on species and environmental surrogates in representing rare, threatened and endemic species and environmental features was evaluated (Chapter 3).

*Combined approach*

The present study uses two combined approaches. The first, similar to that of Bull *et al.* (1993) *in* Margules & Redhead (1995), preselects a set of grids cells (seed points) required to represent all species at least once. From here it then calculates the percentage of each vegetation or landtype already represented in the preselected seed points and then adds on grids cells necessary to ensure that the target levels of each vegetation and landtype representation (5-50%) are reached. This approach will be termed the species-first combined approach. The second approach, or surrogate-first combined approach, preselects a set of grid cells required to represent specified levels of the surrogate classes of vegetation and landtypes (5-50%). The species represented within these preselected grid cells are counted and then grid cells containing the unrepresented species are selected until all species within the database are contained at least once within the set of sites.

These two approaches were then evaluated in terms of efficient representation of regional biodiversity, i.e. maximum biodiversity representation at minimum cost in terms of land required. This efficiency was determined for the species-first approach by calculating the number of additional grid cells required to represent the target levels of the surrogate classes after preselection of the seed points. Similarly, efficiency of the surrogate-first approach was calculated as the number of additional grid cells needed to represent all species at least once after preselection of grid cells containing target levels of the environmental surrogate classes.

*The influence of scale*

The above mentioned analyses use both the environmental surrogates of vegetation types (a broad scale surrogate) and landtypes (a finer scale surrogate). Because the scale and definition of environmental classes can influence the results of studies of this nature (Pressey & Logan, 1995), the effects of the different scales of resolution of these two classes are investigated throughout the present study.

**Results**

*Species based approach*

The species-based conservation area selection algorithm required 116 grid cells (54%) to represent all 2060 species once, more than all conservation areas based on vegetation types, but less than landtype based areas (Figure 1). This conservation area network represents an average of 59% of the 15 vegetation types within the Northern Province (Table 2). Nearly all vegetation types are well represented, with the majority having between 70-80% of their areas represented in the species-based networks. Most lie above a 40% representation with the exception of the Kalahari Plains Thorn Bushveld which is not represented at all and the Lebombo Arid Mountain Bushveld which has only 12.5% of its area represented (Table 2). On average 60% of each of the landtypes are represented in this conservation area network. The majority of landtypes are either not represented at all (0%) or fully represented (100%), the remainder appear to be evenly distributed between all percentage representation classes (Figure 2). Of the 676 landtypes within the Northern Province 134 (20%) are less than 10% represented, while 256 (38%) are more than 90% represented (Figure 2).

*Environmental surrogacy approach*

Figure 3 illustrates the increase in the percentage species captured in conservation areas selected to represent increasing levels of vegetation and landtypes. More than 50% of all species are captured by areas selected to represent vegetation types, however more than 50% of the vegetation type must be selected before 80% of the species are captured, requiring over 45% of the available land area. Areas selected to protect nominated levels of finer-scale landtypes appear to capture more species than areas

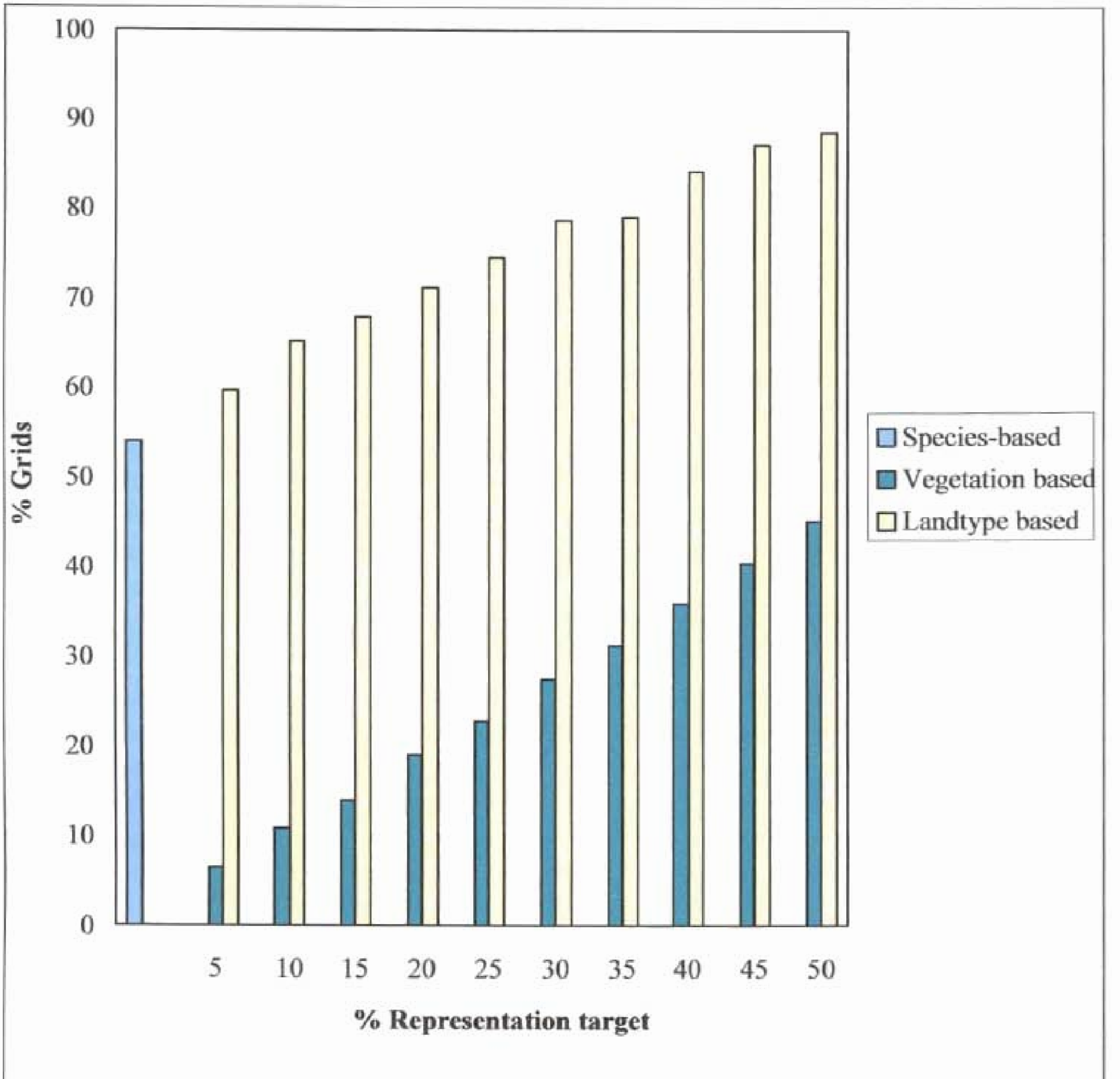


**Figure 1:** Relative efficiencies of conservation area selection procedures measured as the percentage grid cells required.

**Table 2:** Percentage of each vegetation type represented in grid cells selected by species-based approach.

| Vegetation type | % Represented |
|---|---|
| Afromontane Forest | 98.22 |
| Clay Thorn Bushveld | 40.53 |
| Kalahari Plains Thorn Bushveld | 0.00 |
| Lebombo Arid Mountain Bushveld | 12.58 |
| Mixed Bushveld | 49.30 |
| Mixed Lowveld Bushveld | 61.16 |
| Moist Sandy Highveld Grassland | 74.15 |
| Mopane Bushveld | 54.25 |
| Mopane Shrubveld | 77.66 |
| North-eastern Mountain Grassland | 85.97 |
| Sour Lowveld Bushveld | 79.76 |
| Soutpansberg Arid Mountain Bushveld | 77.65 |
| Sweet Bushveld | 34.84 |
| Sweet Lowveld Bushveld | 86.94 |
| Waterberg Moist Mountain Bushveld | 56.11 |

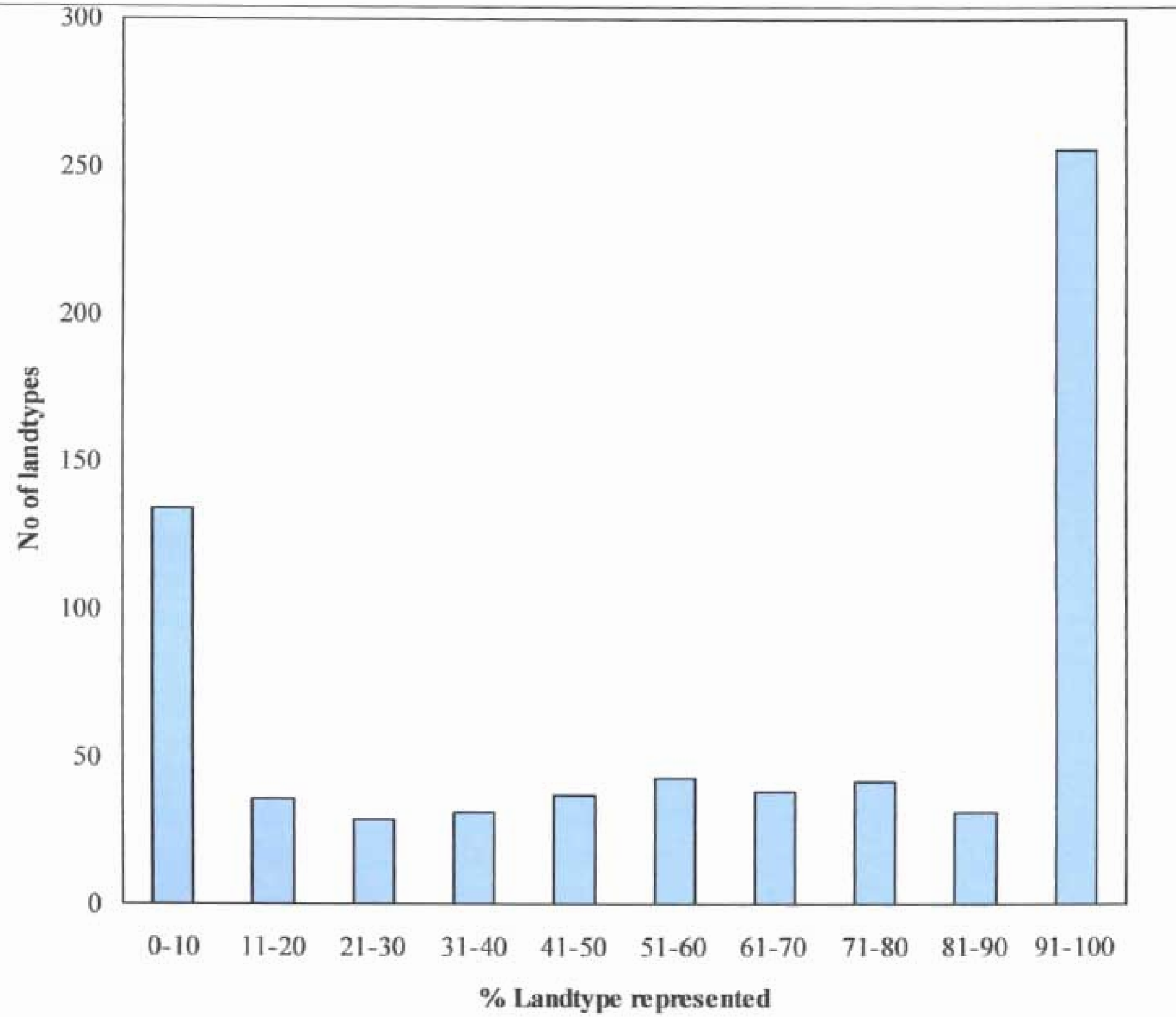


**Figure 2:** Number of landtypes falling into representation classes in a conservation area network based on species data.

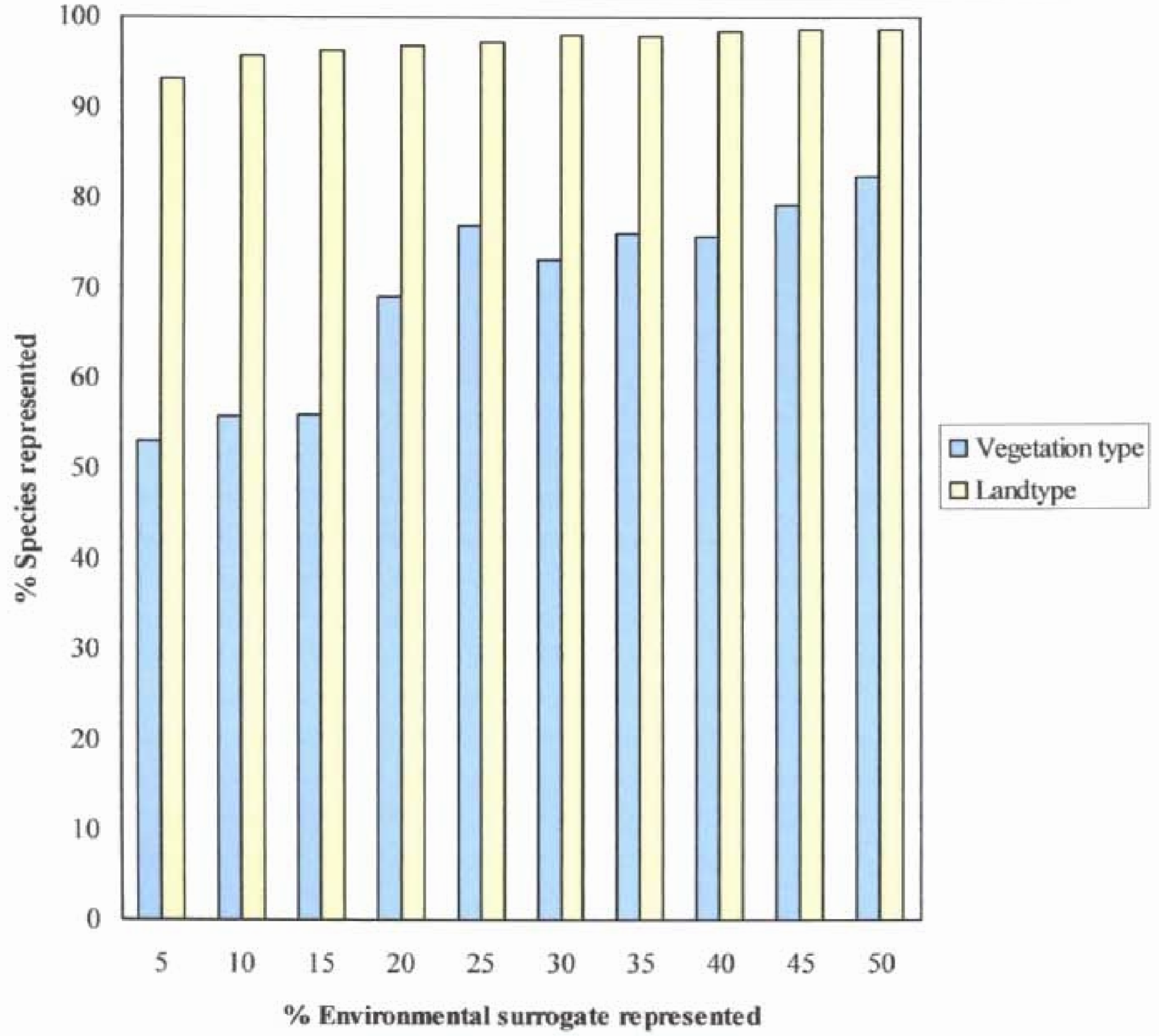


**Figure 3:** Percentage species represented in conservation area networks based on vegetation and landtype representations.

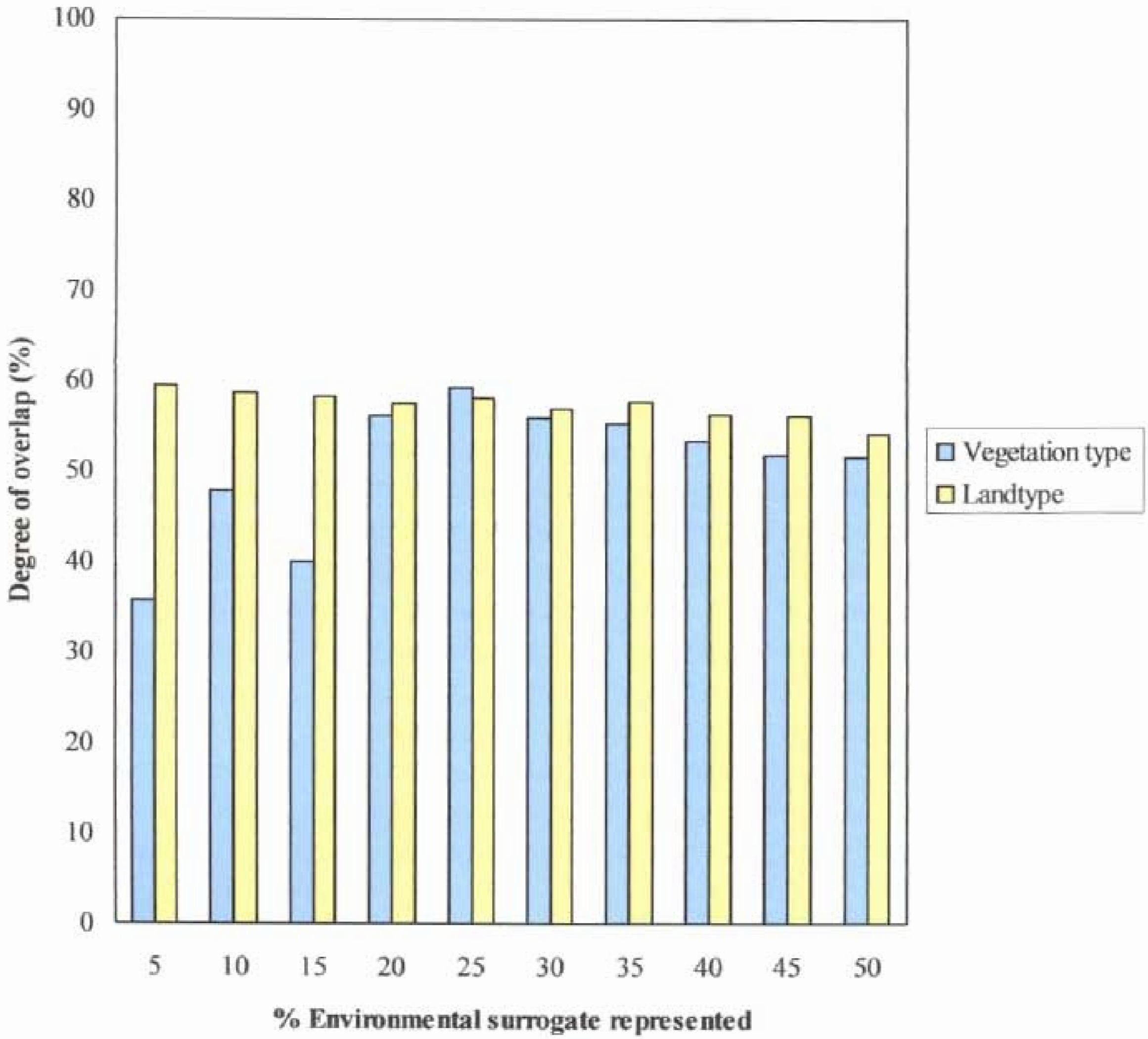


**Figure 4:** Degree of proportional overlap between conservation area networks based on species distribution data, and vegetation and landtype representations.

The landtype based conservation areas capture species well, representing over 90% of species within the region when just five percent of the landtypes being represented. However this comes at a high cost to land, requiring almost 60% of the land area. Although the number of species increases with increasing levels in surrogate representation, this increase requires a disproportionate increase in land area. An increase of 30% in species represented by the vegetation based areas (from 52-82%) requires an almost 40% increase in land (from 7-45%). While a five percent increase (from 93-98%) in species captured by landtype based conservation areas requires an almost 30% (from 59-88%) increase in land area.

*Comparison of approaches*

The success at which each of the species and environmental surrogate approaches represent non-target biodiversity features (species, vegetation and landtypes) is varied. Species-bases approaches represent relatively high levels of vegetation types and landtypes, while environmental surrogate approaches also represent species well, but at a high cost to land. The spatial configurations of these different sets of conservation areas, compared using measures of proportional overlap (Figure 4), demonstrate a relatively low degree of overlap and suggest that areas of conservation importance to species do not necessarily coincide with areas identified for the efficient representation of vegetation and landtypes.

Figure 5 illustrates the success with which the environmental surrogate based approaches capture species important to effective conservation, i.e. rare and endemic species. Landtype based conservation areas are very effective at representing rare and endemic species, representing between 89 and 98% of all rare and endemic species identified. While the vegetation based areas are not as effective, especially at low levels of vegetation types representation, only representing over 50% of the rare and endemic species when more than 20% of each vegetation types is represented. Once all vegetation types are 50% represented, still more than 30% of these important species are excluded.

*Combined approach*

Figure 6 illustrates the results of the species and environmental surrogate first approaches. In the species first approach (Figures. 6a & 6b) there is a general increase in the number of additional grid cells required as the percentage of vegetation and landtypes represented increase. The number of additional grid cells required to protect a specified percentage of landtypes (Figure 6b) is more than that required to protect the same level of vegetation types (Figure 6a). Figures 6c and 6d show the number of grid cells required for the surrogate-first approach based on vegetation and landtypes respectively. More additional grid cells are required to protect all species when vegetation types (Figure 6c) are pre-selected than when landtypes are pre-selected (Figure 6d). However, the vegetation types require fewer initial grid cells than the landtypes, thus the vegetation type based approach uses fewer grid cells in total for this surrogate-first approach. As both vegetation and landtype based approaches tend towards a 50% level of representation, so the number of additional grid cells required decreases.

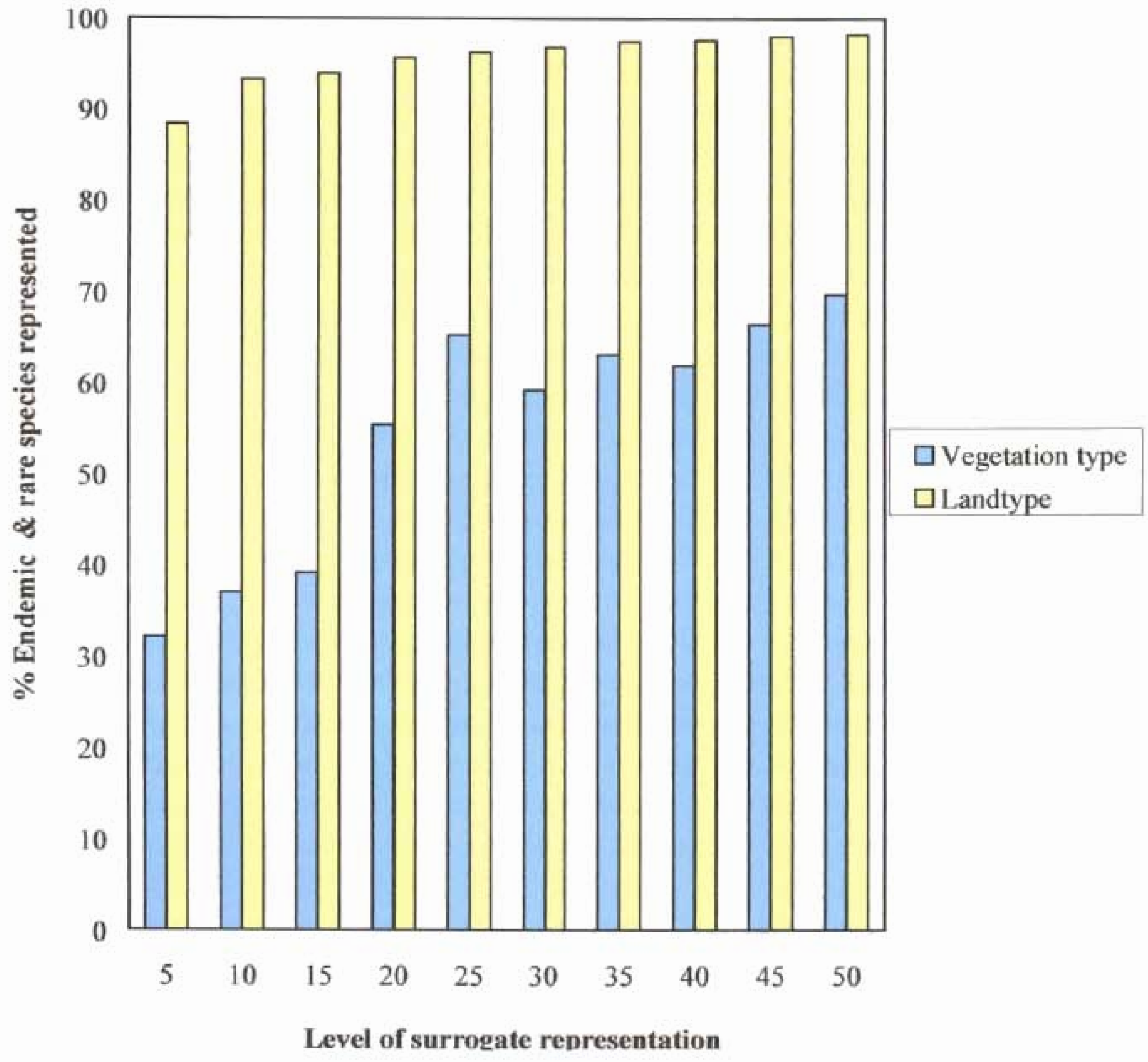


**Figure 5:** Percentage rare and endemic species captured by vegetation type and landtype based conservation areas.

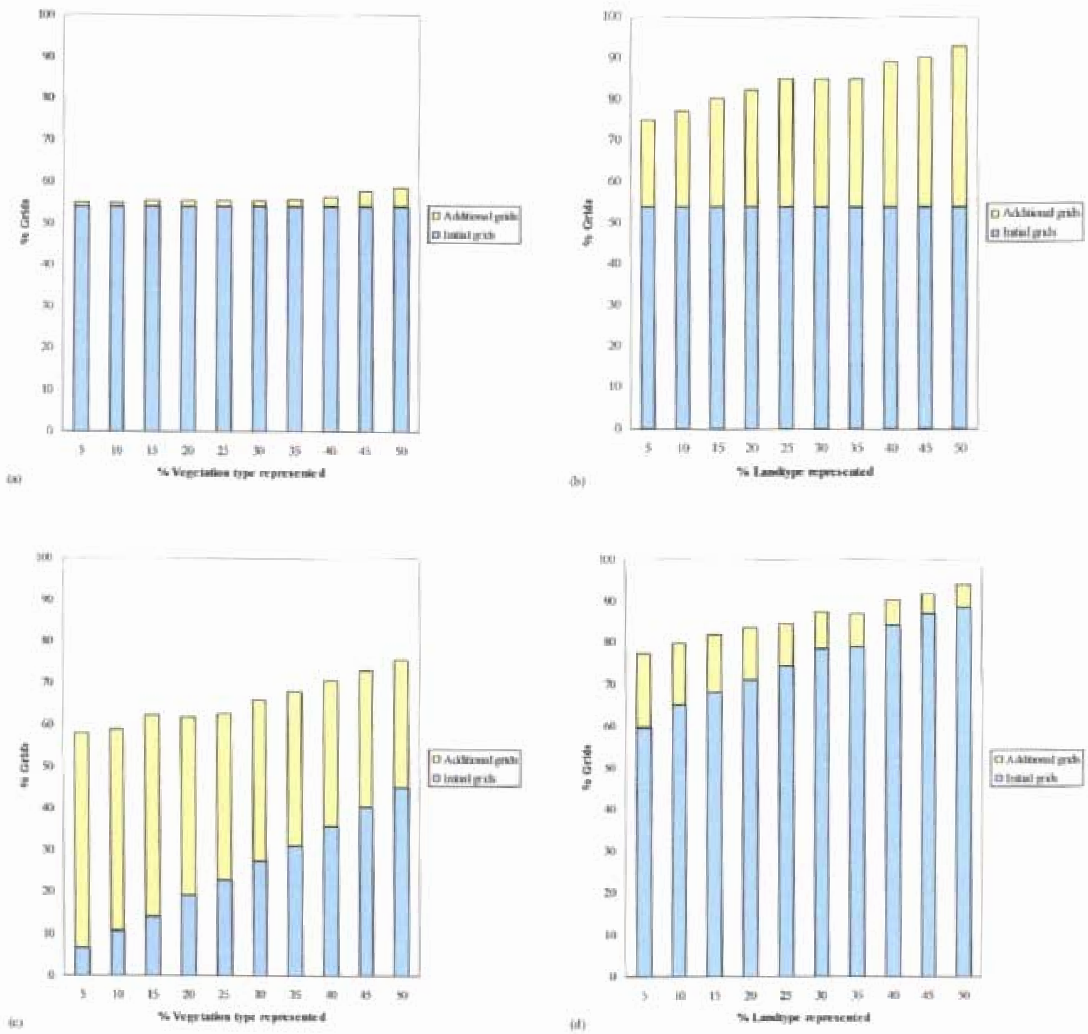


**Figure 6:** Percentage of initial and additional grid cells required to represent: (a) a required percentage of each vegetation type; and (b) landtype using a species first approach to conservation area selection; and (c) all species using a vegetation type first; and (d) landtype first approach to conservation area selection.

**Discussion**

Although many studies using surrogate measures of biodiversity in an attempt to identify areas important to the conservation of regional biodiversity have been conducted, they have made little attempt to test whether a relationship exits between these surrogate classes and biodiversity within the area (Pressey & Nicholls, 1989b; Bedward *et al.*, 1992; Pressey & Tully, 1994). Many authors have actually questioned the existence of such a relationship and recommend that it be demonstrated before being applied in conservation planning decision making (Landres *et al.*, 1988; Bedward *et al.*, 1992; Pressey, 1994; Wessels *et al.*, 1999). This study highlights various aspects of this assumed relationship. First, it is once again evident from the results that support for the use of species or environmental measures as surrogates for regional biodiversity will depend on the assessment techniques used (Chapter 3), with measures of proportional overlap showing little support while levels of non-target feature representation provide more support for surrogate measures.

Second, the more effective the conservation area selection techniques are at representing regional biodiversity, the less land-use efficient they become, requiring large tracts of land. This trade-off between the degree of feature representation achieved within surrogate based conservation areas and the amount of land required is a recurrent theme in conservation planning (Chapters 3) (Williams & Humphries, 1996; Pressey & Logan, 1995; Pressey & Logan, 1998; Wessels *et al.*, 1999). Similarly the scale at which the surrogate classes are defined, the number of these classes and the size of the selection units will also influence the outcome and efficiency (Bedward *et al.*, 1992; Nicholls & Margules, 1993; Pressey & Logan, 1995). With larger or more classes (e.g. landtype classes) and selection units often resulting in overrepresentation of regional biodiversity features and a decrease in land-use efficiency. This however is often traded off against an improvement in the persistence of organisms in larger reserves and a smaller need for expensive interventionist management (Pressey & Logan, 1998). The more heterogeneous classes of vegetation types do seem to miss some of the underlying variation in species diversity, which the finer homogenous landtype classes seem to capture. However, it is difficult to make definite conclusions on this aspect of scale due to the difference in total area required by the two approaches, with landtypes requiring much more land to reach the same levels of surrogate representation as vegetation types (Fig. 7).

Figure 7 illustrate this trade off between effective biodiversity representation and efficient land-use for conservation area selection by species and environmental surrogate approaches, both separately and combined. It illustrates the higher land-use requirements when using a surrogate like landtypes which have many classes defined at a finer resolution, as opposed to the vegetation types. The vegetation type approach (Veg) requires little land but achieves low levels of species and environmental surrogates representation, whereas the combined approaches using landtype data (Comsplan and Comlan) represent many species but at a much higher land-use cost.

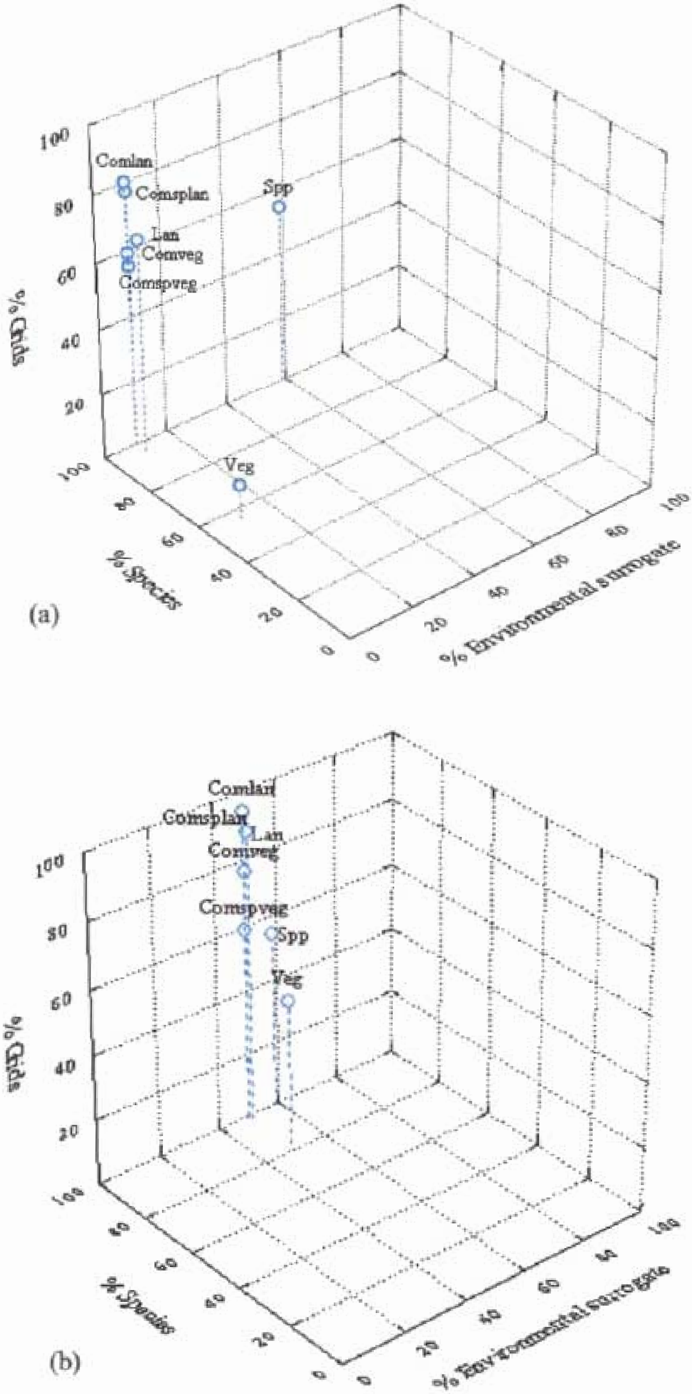


**Figure 7:** Graphical comparison of the efficiencies (% grid cells required) and representativeness (% species and % environmental surrogates represented) of conservation area selection techniques targeting (a) 10%; and (b) 50% surrogate representation. Techniques include: species-based (**Spp**), vegetation type based (**Veg**), landtype based (**Lan**), combined species first with vegetation type targets (**Comspveg**), combined species first with landtype targets (**Comsplan**), combined vegetation type first with species targets (**Comveg**), and combined landtype first with species targets (**Comlan**).

It appears that both species first and environmental surrogate first combined approaches display similar degrees of success in representing species and environmental surrogates, but that the landtype based ones are more costly in terms of land area. It therefore appears that the species-based approach (Spp) effectively reaches a compromise between representing many species and moderately high levels of environmental surrogates at a lower cost to land. It must however be noted that levels of environmental surrogates represented by the environmental surrogate based techniques in Figure 7 are averages of the representation target levels (5-50%).

Third, although the results appear encouraging for the use of both species and environmental surrogates in conservation area selection in that they capture many non-target biodiversity features, these approaches still exclude some important components of regional biodiversity. As found in Chapter 3 as well as other studies on biodiversity surrogates (Prendergast *et al.*, 1993; Curnutt *et al.*, 1994; Williams *et al.*, 1996; Dobson *et al.*, 1997), many surrogate approaches miss species and other biodiversity features of conservation importance due to high levels of threat, endemism or rarity. The vegetation type approaches miss many rare and endemic species, while the landtype approach selects many more grid cells and misses less of these species. The species-based approach may represent high levels of vegetation and landtypes, however it does exclude some of these totally in the areas identified.

This approach excludes nearly 20% of all the landtypes and although it represents the vegetation types well, the Kalahari Plains Thorn Bushveld it totally excluded and is recognised as one of the most threatened vegetation types in South Africa (see Addendum I). This is an important shortcoming in most current biodiversity surrogate measures and must be highlighted. As Pressey (1994) points out, it is not only the geographic rarity and the increased likelihood of being missed by conservation areas that makes threatened, rare and endemic biodiversity features a conservation priority. Even if some of these features are captured in the coarse-filter surrogate approach they will not necessarily be adequately protected, often requiring additional protection and active management.

Lastly, these results seem to suggest that the 10% protected area coverage recommended by the IUCN (1993) is far from adequate. Protecting 10% of all vegetation types only represents some 50% of the species known to occur in the Northern Province, and excludes almost 65% of all rare and endemic species. Thus this study supports Soulé and Sanjayan (1998) in their review of findings on conservation targets where they illustrate that approximately 50% of the land area would be required to represent and protect most elements of biodiversity. Not only are these politically convenient conservation targets therefore inadequate in preventing a mass extinction of species, they also run the risk of becoming ceilings above which no nation feels the need to protect.

Thus it would appear from the results that the best approach to conservation area selection is one that uses all available forms of data, thereby incorporating more biodiversity components (Lombard *et al.*, 1997; Maddock & Du Plessis, 1999; Maddock & Benn, 2000). Including both species and environmental measures into selection procedures ensures that not only are all facets of biodiversity better represented but that the important aspects of biodiversity; the threatened, endemic and rare features, are also captured. This of course requires much land area and thus perhaps off-reserve management and conservation are the only feasible ways of ensuring that these identified areas are guaranteed some level of protection, even if it is outside of reserves (Pressey & Logan, 1997).

**References**

Austin, M.P. & Margules, C.R. (1986). Assessing representativeness. In *Wildlife conservation evaluation*. (Ed. M.B. Usher), pp 45-67. Chapman and Hall, London.

Balmford, A. (1998). On hotspots and the use of indicators for reserve selection. *Trends in Ecology & Evolution* 13, 409.

Bedward, M., Pressey, R.L. & Keith, D.A. (1992). A new approach for selecting fully representative reserve networks: addressing efficiency, reserve design and land suitability with an iterative analysis. *Biological Conservation* 62, 115-125.

Belbin, L. (1993). Environmental representativeness: regional partitioning and reserve selection. *Biological Conservation* 66, 223-230.

Bull, A.L., Thackway, R. & Cresswell, I.D. (1993*). Assessing conservation of the major Murray-Darling basin ecosystems*. Environmental Resources Information Network (ERIN), Australian Nature Conservation Agency, Canberra.

Christian, C.S. & Steward, G.A. (1968). Methodology of integrated surveys. In *Proceedings of the Toulouse conference on aerial and integrated studies,* pp 233-280. UNESCO, Paris.

Church, R. L. Stoms D. M. & Davis F. W. (1996). Reserve selection as a maximal covering location problem. *Biological Conservation* 76, 105-112.

Cowling, R.M. & Heijnis, C.J. (*In Press*). The identification of broad habitat units as biodiversity entities for systematic conservation planning in the Cape Floristic Region. *South African Journal of Botany*

Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L., Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Manuela, H. & Sahr, K. (1997). A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80, 83-97.

Curnutt, J., Lockwood, J., Luh, H.K., Nott, P. & Russell, G. (1994). Hotspots and species diversity. *Nature* 367, 326-327.

Dobson, A.P., Rodriguez, J.P., Roberts, W.M. & Wilcove, D.S. (1997). Geographic distribution of endangered species in the United States. *Science* 275, 550-553.

Fairbanks, D.H.K. & Benn, G.A. (2000). Identifying regional landscapes for conservation planning: A case study from KwaZulu-Natal, South Africa. *Landscape and Urban Planning* 50, 1-19.

Faith, D.P. & Norris, R.H. (1989). Correlation of environmental variables with patterns of distribution and abundance of common and rare freshwater macroinvertebrates. *Biological Conservation* 50, 77-99.

Faith, D.P. & Walker, P.A. (1996). How do indicator groups provide information about the relative biodiversity of different sets of areas?: on hotspots, complementarity and pattern-based approaches. *Biodiversity Letters* 3, 18-25.

Flather, C.H., Wilson, K.R., Dean, D.J. & McComb, W.C. (1997). Identifying gaps in conservation networks: of indicators and uncertainty in geographic-based analyses. *Ecological Applications* 7, 531-542.

Freitag, S., Hobson, C., Biggs, H.C. & van Jaarsveld, A.S. (1998). Testing for potential survey bias: the effect of roads, urban areas and nature reserves on a Southern African mammal data set. *Animal Conservation* 1, 119-127.

Freitag, S. & Mansell M.W. (1997). The distribution and protection status of selected antlion species (Neuroptera: Mymeleontidae) in South Africa. *African Entomology* 5, 205-16.

Freitag, S., Nicholls, A.O. & van Jaarsveld, A.S. (1996). Nature reserve selection in the Transvaal, South Africa: what data should we be using? *Biodiversity Conservation* 5, 685-698.

Freitag, S. & van Jaarsveld, A.S. (1995). Towards conserving regional mammalian species diversity: a case study and data critique. *South African Journal of Zoology* 30, 136-144.

Freitag, S., van Jaarsveld, A.S. & Bigss, H.C. (1997). Ranking priority areas: an iterative conservation value based approach. *Biological Conservation* 82, 263-272.

Gaston, K.J. (1996). Species richness: measure and measurement. In *Biodiversity: a biology of numbers and difference*. (Ed. K.J. Gaston), pp. 77-113. Blackwell Science, Oxford.

Gaston, K.J. & Williams, P.H. (1993). Mapping the world's species - the higher taxon approach. *Biodiversity Letters* 1, 2-8.

Haila, Y. & Margules, C.R. (1996). Survey research in conservation biology. *Ecography* 19, 323-331.

Harrison, J.A. (1992). The South African Bird Atlas Project: Five years of growth. *South African Journal of Science* 88, 410-413.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W., Baltzer, M., Dickinson, C.J., Lwanga, J.S., Matthews, R.A. & Balmford, A. (1998). Complementarity and the use of indicator groups for reserve selection in Uganda. *Nature* 394, 472-475.

Hull, H.E., Chown, S.L. & Bellamy, C.L. (1998). Identification and evaluation of priority vonservation areas for Buprestidae (Coleoptera) in South Africa, Lesotho, Swaziland and Namibia. *African Entomology* 6, 265-274.

IUCN. (1993). *Parks for life: report of the IVth world congress on national parks and protected areas*. IUCN, Gland, Switzerland.

Kerr, J.T. (1997). Species richness, endemism, and the choice of areas for conservation. *Conservation Biology* 11, 1094-100.

Kirkpatrick, J.B. (1983). An iterative method for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation* 25, 127-134.

Koch, S.O., Chown, S.L., Davis, A.L.V., Endrödy-Younga, S. & van Jaarsveld, A.S. (2000). Conservation strategies for poorly surveyed taxa: a dung beetle (Coleoptera, Scarabaeidae) case study from southern Africa. *Journal of Insect Conservation* 4, 45-56.

Land Type Survey Staff. (1986). Land types of the map 2730 Vryheid. *Memoirs of Agricultural Natural Resouces of South Africa* No. 7.

Landres, P.B., Verner, J. & Thomas, J.W. (1988). Ecological uses of vertebrate indicator species: a critque. *Conservation Biology* 2, 316-328.

Launer, A.E. & Murphy, D.D. (1994). Umbrella species and the conservation of habitat fragments: a case of a threatened butterfly and a vanishing grassland ecosystem. *Biological Conservation* 69, 145-153.

Lawrance, C.J., Byard, R.J. & Beavan, P.J. (1993). *Terrain Evaluation Manual*. HMSO, London.

Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larsen, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S. & Watt, A.D. (1998) Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. *Nature* 391, 72-75.

Lombard, A.T. (1995). The problem with multi-species conservation: do hotspots, ideal reserves and existing reserves coincide? *South African Journal of Zoology* 30, 145-163.

Lombard, A.T., Cowling, R.M., Pressey, R.L. & Mustart, P.J. (1997). Reserve selection in a species-rich and fragmented landscape on the Augulhas plain, South Africa. *Conservation Biology* 11, 1101-1116.

Lombard, A.T., Nicholls, A.O., & August, P.V. (1995). Where should nature reserves be located in South Africa? A snake's perspective. *Conservation Biology* 9, 363-372.

Low, A.B. & Rebelo, A.G. (1996). *Vegetation of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs and Tourism, Pretoria.

Mace, G.M., Balmford, A., Buitani, L., Cowlishaw, G., Dobson, A.P., Faith, D.P., Gaston, K.J., Humphries, C.J., Vane-Wright, R.I., Williams, P.H., Lawton, J.H., Margules, C.R., May, R.M., Nicholls, A.O., Possingham, H.P., Rahbek, C. & van Jaarsveld, A.S. (2000). Its time to work together and stop duplication conservation efforts. *Nature* 405, 393.

MacVicar, C.N., Scotney, D.M., Skinner, T.E., Niehaus, H.S. & Loubser, J.H. (1974). A classification of land (climate, terrain form, soil) primarily for rainfed agriculture. *South African Journal of Agriculture Extensions* 3, 21-24.

Maddock, A. & Benn, G.A. (2000). Identification of conservation-worthy areas in northern Zululand, South Africa. *Conservation Biology* 14, 155-166.

Maddock, A. & Du Plessis, M.A. (1999). Can species data only be appropriately used to conserve biodiversity. *Biodiversity & Conservation* 8, 603-615.

Margules, C.R., Cresswell, I.D. & Nicholls, A.O. (1994). A scientific basis for establishing networks of protected areas. In *Systematics and Conservation Evaluation*. (Eds. Forey, P.L., Humphries, C.J.& Vane-Wright, R.I.), pp 327-350. Clarendon Press, Oxford.

Margules, C.R., Nicholls, A.O. & Pressey, R.L. (1988). Selecting networks of reserves to maximise biological diversity. *Biological Conservation* 43: 63-76.

Margules, C.R. & Pressey, R.L. (2000). Review article: Systematic conservation planning. *Nature* 405, 243-253.

Margules, C. R. & Redhead, T.D. (1995). *BioRap: guidelines for using the BioRap methodology and tools*. CSIRO, Australia.

Muller, C. (1999). *The distribution and conservation of termites (Isoptera) and butterflies (Lepidoptera) in South Africa*. Unpublished MSc dissertation: University of Pretoria, Pretoria, South Africa.

Muller, C., Freitag, S., Scholtz, C.H. & van Jaarsveld, A.S. (1997). Termite (Isoptera) distributions, endemism, species richness and priority conservation areas: consequences for land-use planning in South Africa. *African Entomology* 5, 261-271.

Nicholls, A.O. & Margules, C.R. (1993). An upgraded reserve selection algorithm. *Biological Conservation* 64, 165-169.

Noss, R.F. (1983). A regional landscape approach to maintaining diversity. *BioScience* 33, 700-706.

Noss, R.F. (1987). From plant communities to landscapes in conservation inventories: A look at The Nature Conservancy (USA). *Biological Conservation* 84, 223-234.

Panzer, R. & Schwartz, M.W. (1998). Effectiveness of a vegetation-based approach to insect conservation. *Conservation Biology* 12, 693-702.

Pimm, S. L. & Raven P. (2000). Extinction by numbers. *Nature 403*: 843-845.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. (1993). Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature* 356, 335-337.

Pressey, R.L. (1994). Land classifications are necessary for conservation planning but what do they tell us about fauna. In *Future of the Fauna of Western New South Wales*. (Eds. D. Lunney, S. Hand, P. Reed & D. Butcher), ch 3. The Royal Zoological Society of New South Wales, Australia.

Pressey, R.L., Hager, T.C., Ryan, K.M., Schwarz, J., Wall, S., Ferrier, S. & Creaser, P.M. (2000). Using abiotic data for conservation assessments over extensive regions: quantitative methods applied across New South Wales, Australia. *Biological Conservation* 96, 55-82.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I., & Williams, P.H. (1993). Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology & Evolution* 8, 124-128.

Pressey, R.L. & Logan V.S. (1995). Reserve coverage and requirements in relation to partitioning and generalisation of land-classes: analyses for Western New South Wales. *Conservation Biology* 9, 1506-1517.

Pressey, R.L. & Logan, V.S. (1997). Inside looking out: Findings of research on reserve selection relevant to "off-reserve" nature conservation. In *Conservation Outside Nature Reserves*. (Eds. P. Hale & D. Lamb), pp. 407-418. Centre for Conservation Biology, University of Queensland.

Pressey, R.L. & Logan V.S. (1998). Size of selection units for future reserves and its influence on actual vs targeted representation of features: a case study in western New South Wales. *Biological Conservation* 85, 305-319.

Pressey, R.L. & Nicholls, A.O. (1989a). Efficiency in conservation evaluation: scoring versus iterative procedures. *Biological Conservation* 50, 199-218

Pressey, R.L. & Nicholls, A.O. (1989b). Application of a numerical algorithm to the selection of reserves in semi-arid New South Wales. *Biological Conservation* 50, 264-278.

Pressey, R.L. & Tully, S.L. (1994). The cost of ad hoc reservation: A case study in western New South Wales. *Australian Journal of Ecology* 19, 375-384.

Purdie, R.W., Blick, R. & Bolton, M.P. (1986). Selection of a conservation reserve in the Mulga biogeograpgic region of south-western Queensland, Australia. *Biological Conservation* 38, 369-384.

Rebelo, A.G. & Siegfried, W.R. (1992). Where should nature reserves be located in the Cape floristic region, South Africa? Models for the spatial configuration of a reserve network aimed at maximizing the protection of floral diversity. *Conservation Biology* 6, 243-252.

Reid, W. V. (1998). Biodiversity hotspots. *Trends in Ecology & Evolution* 13, 275-280.

Scott, J.M., Csuti, B., Estes, J.E. & Anderson, H. (1989). Status assessment of biodiversity protection. *Conservation Biology* 3, 85-87.

Scott, J.M., Csuti, B., Jacobi, J.D. & Estes, J.E. (1987). Species richness: a geographic approach to protecting future biological diversity. *BioScience* 37, 782-788.

Soulé, M.E. & Sanjayan, M.A. (1998). Conservation targets: do they help? *Science* 279, 2060-2061.

Underhill, L.G. (1994). Optimal and suboptimal reserve selection algorithms. *Biological Conservation* 70, 85-87.

UNEP. (1995). *Global Biodiversity Assessment*. Cambridge University Press.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998). Biodiversity assessments and conservation strategies. *Science* 279, 2106-2108.

Wessels, K.J., Freitag, S. & van Jaarsveld, A.S. (1999). The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological Conservation* 89, 21-38.

Williams, P.H. & Gaston, K.J. (1994_ Measuring more of biodiversity: can higher taxon richness predict wholesale species richness? *Biological Conservation* 67, 211-217.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C. & Pressey, R. (1996). A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. *Conservation Biology* 10, 155-174.

Williams, P.H. & Humphries, C.J. (1996). Comparing character diversity among biotas. In *Biodiversity: a biology of numbers and difference.* (Ed. K.J. Gaston), pp 54-76. Blackwell Science, Oxford.

Williams, P.H. Humphries, C.J. & Gaston K J. (1994). Centres of seed-plant diversity: the family way. *Proceedings of the Royal Society, London (B)* 256, 67-70.

Willis, C.K., Lombard, A.T., Cowling, R.M., Heydenrych, B.J. & Burgers, C.J. (1996). Reserve systems for limestone endemic flora of the Cape lowland fynbos: iterative versus linear programming. *Biological Conservation* 77, 53-62.

Wilson, E.O. (1988). The current state of biological diversity. In *Biodiversity*. (Ed. E.O. Wilson), pp 3-20. National Academic Press, Washington.

# CHAPTER 5

## A multicriteria approach to reserve selection: addressing long-term biodiversity maintenance

# A multicriteria approach to reserve selection: addressing long-term biodiversity maintenance

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**Abstract**

Existing reserve selection techniques concerned with maximal biodiversity representation within minimum land area, do not necessarily ensure the long-term maintenance of biodiversity. These traditional approaches ignore the maintenance of natural processes, turnover of feature diversity and the need to minimise threats within conservation areas. We address these three emergent issues in the identification of potential avian conservation areas in the Northern Province of South Africa, by combining ordination and spatial autocorrelation analyses, as well as land transformation data into traditional reserve selection techniques. Existing conservation areas are biased and inefficient and traditional methods do little to correct this skew. The inclusion of species assemblage structure as well as the underlying environmental gradients ensures a conservation area network that strives to maintain both biodiversity pattern and process. Spatial autocorrelation analysis allows for the identification of areas with high beta diversity, important areas for the long-term maintenance of biodiversity. The inclusion of land transformation data leads to viable conservation area networks and highlights areas of potential conflict between biodiversity conservation interests and human land-use issues. These improvements on the current generation of reserve selection techniques bring us a step closer to ensuring the long-term maintenance of biodiversity within conservation areas.

**Introduction**

Historic opportunistic methods for assigning land with low potential for economic and political conflict; or high potential for recreation and tourism to biodiversity conservation are inefficient and ultimately more costly means of conservation area allocation (for review see Pressey & Tully, 1994; Rodrigues *et al.*, 1999). This realisation has led to the development of a wide array of systematic reserve selection techniques designed to make conservation area selection more efficient. These techniques are concerned with the maximal coverage problem and include complementarity-based reserve selection and linear programming algorithms (for review see Church *et al.*, 1996; Csuti *et al.*, 1997; Williams, 1998; Margules & Pressey, 2000). However, the ultimate goal of conservation planning is to ensure the long-term security of the planet's biodiversity (Scott *et al.*, 1989) and the extent to which conservation areas fulfil this role depends only partly on this long established representation goal concerned mainly with sampling the biodiversity pattern in a complementary fashion.

Several authors have emphasised that current biodiversity representation within conservation areas is not equivalent to the ultimate goal of maintaining biodiversity in the long-term (Balmford *et al.*, 1998; Williams, 1998; Cowling *et al.*, 1999; Margules & Pressey, 2000; Rodrigues *et al.*, 2000). Many suggestions have been put forward as to how to ensure the long-term maintenance of biodiversity within conservation areas. Williams (1998) suggests promoting viability and reducing threat as vital components of the conservation area selection process. Margules and Pressey (2000) agree but add the maintenance of natural processes as an important component of conservation area selection. Cowling *et al.* (1999) separate these issues into retention goals, generally formulated in the context of threats to biodiversity, and long-term persistence goals, which concern the maintenance of natural processes. Rodrigues *et al.* (2000) argue that as species diversity distribution patterns change over time, the selection of conservation areas which are robust to turnover in feature diversity is a critical component of conservation area selection for ensuring the long-term maintenance of biodiversity. We address all three emergent issues of natural process maintenance, turnover of feature diversity and minimising threat in the present study as we identify potential avian conservation areas in the Northern Province of South Africa for the long-term maintenance of regional avian diversity.

*Maintenance of natural processes*

In order to ensure the long-term maintenance of biodiversity within conservation areas, these areas must conserve not only the biodiversity pattern, but also the natural processes that control and maintain that pattern (Balmford *et al.*, 1998). Conservation of ecosystem processes that sustain ecosystem structure and function, and evolutionary processes that sustain lineages and generate diversity, are essential for achieving the long-term maintenance of biodiversity in conservation areas (Nicholls, 1998). These processes include interspecific interactions, regular and nomadic faunal movements, disturbance regimes and climate change among many others (Balmford *et al.*, 1998; Cowling *et al.*, 1999). But, as Margules and Pressey (2000) point out, because conservation area selection is often a spatial exercise, protection of these natural processes is often based on their spatial surrogates rather then on the processes themselves. These surrogates can include size, lack of roads, watershed boundaries, dispersal routes, land classes, landscapes and other geographic features (Cowling *et al.*, 1999; Margules & Pressey, 2000). There are numerous natural processes that need to be considered and, as is often the case, they have not been adequately documented for a region such as the Northern Province. However, as Noss (1996) points out, by ensuring that conservation areas are large and span substantial environmental gradients it should be possible to accommodate, at least partially, many of these processes.

We aim therefore to investigate not only the avian diversity pattern within the region but also the processes responsible for that pattern by identifying broad avian communities in the study area, as well as the environmental gradients that drive this emergent community structure. In pursuit of identifying these environmental gradients responsible for the biodiversity pattern, ordination (gradient) analyses have illustrated tremendous potential (Taggart, 1994; Faith, 1995; Faith & Walker, 1996a). This analytical approach has a long history in community ecology and is used for identifying key processes responsible for the control and maintenance of biodiversity patterns by integrating multiple environmental effects across a landscape (Bray & Curtis, 1957; Whittaker & Niering, 1965; Jongman *et al.*, 1995). In addition to this, ordination explores biodiversity patterns by interpreting entire species assemblages of sample sites, rather than through species-by-species relations, offering community level results and responses to often complex environmental gradients (Jongman *et al.*, 1995).

*Turnover in feature diversity*

Despite the importance of beta diversity in determining regional species richness patterns, little attention has been paid to this component of diversity within conservation area selection (Whittaker, 1977; Cowling *et al.*, 1989). Alpha diversity refers to the number of species within a homogenous community (Whittaker, 1972; 1977); beta diversity on the other hand is concerned with species turnover or the rate at which species are replaced by others along habitat gradients (Whittaker, 1972). Conservation areas selected based on representation of alpha feature diversity patterns, without considering the turnover of features or beta diversity, may not necessarily continue to serve their purpose over a period of years. Several authors have found that traditional complementary-based approaches to conservation area selection may not be adequate if the role of a conservation areas is to maintain biodiversity in the long-term rather than simply represent current biodiversity patterns (Margules *et al.*, 1994; Virolainen *et al.*, 1999; Rodrigues *et al.*, 2000). In this study we aim to address this issue of beta diversity by identifying and focussing on areas with a high turnover in species along associated environmental gradients.

*Minimising threat*

The basic role of conservation areas is to protect elements of biodiversity from external processes and factors that threaten their existence (Margules & Pressey, 2000). Very few of the existing methods for identifying conservation areas include measures of threat into the selection process (Balmford *et al.*, 1998; Faith & Walker, 1996b; Williams, 1998). Land-cover changes, caused mainly by agriculture and urban development, present the single most important threat to global biodiversity (Soulé, 1991; Dale *et al.*, 1994). As a result, signatories to the Convention on Biological Diversity are obligated to assess the impact of land transformation on biodiversity and to implement appropriate responses if necessary (DEAT, 1997).

Many areas identified as important areas for conservation based on historical species distribution data may in reality be largely transformed (Wessels *et al.*, 2000 (see Addendum II)). In addition, although transformed areas may currently harbour some species, these areas may not be able to sustain natural ecological processes and complete samples of non-target taxa (Baudry, 1993; Di Benedetto *et al.*, 1993; Freemark, 1995), thus precluding these areas from viable conservation area networks. Therefore the incorporation of land-cover (the suite of natural and human-made features that cover the earth's immediate surface) information into conservation planning is essential (Wessels *et al.*, 2000 (see Addendum II)). We include these data into conservation area selection techniques in the Northern Province of South Africa in order to identify and minimise threats within the proposed avian conservation area network.

## Methods

### *Study area*

The Northern Province of South Africa occupies about 10% (122305 km$^2$) of the country and lies at the northeastern tip of South Africa bordering on the countries of Mozambique, Botswana and Zimbabwe (Figure 1). The province includes the northern end of the Drakensberg escarpment which separates the low-lying, warm and humid Lowveld region on the east from the higher lying, drier and cooler Bushveld plateau region in the west (Figure 1). The Limpopo river forms the northern and northeastern boundary of the province where it borders on the neighbouring states of Botswana and Zimbabwe. This Limpopo river valley is separated from the Lowveld and central Bushveld plateau by the Soutpansberg and Blouberg mountain ranges. The Waterberg mountain range falls within the central Bushveld plateau region and together with the escarpment encircles the Springbok flats, a clay substrate basin within the Bushveld plateau with a long history of dry land cultivation (Figure 1).

The study area consists primarily of the savanna biome, with small areas on the escarpment covered by grassland and forest biomes (Low & Rebelo, 1996). The province includes extensive areas of arable land and as a result 14% of the province has been transformed by cultivation. Urbanisation (1.6%) and forestry plantations (0.8%) account for the remaining land transformations (Thompson, 1996; Fairbanks *et al.*, 2000). However the study area has not been excessively degraded and transformed since 73% is still covered by natural vegetation (Table 1; Figure 2) and 11.36% is under formal protection in provincial and national protected areas.

*Avian distribution data*

Information on avian distribution at a quarter degree grid cell (15' x 15'; ~700 km$^2$ hereafter referred to as a grid cell) resolution was collated from the South African Bird Atlas Project (Harrison, 1992; Harrison *et al.*, 1997). The presence/absence of 565 avian species, comprising 60% of the bird diversity recorded in the Southern African sub-region (South Africa, Lesotho, Swaziland, Namibia, Botswana, Zimbabwe and southern Mozambique), was recorded from 1980–1992. The Northern Province contains 48803 unique distribution records for these bird species within 214 grid cells (Figure 3).

*Environmental data*

Among the factors and processes that have been hypothesised to account for spatial patterns of species diversity are climatic extremes, climatic stability, productivity, and habitat heterogeneity (Brown, 1995; Wickham *et al.*, 1997). Data were compiled from existing sources to represent these factors (Table 2), including interpolated weather stations (Schulze, 1998) and topographic contours (SA Surveyor General, 1993) mapped in a geographic information system (GIS; ESRI, 1998) using Albers equal area projection. This GIS database had a grid cell resolution of 1km x 1km, which was determined by the cell size of existing rasterised data sets and a logical cell size for future integrative work.

*Land-cover data*

Land-cover data were mapped from 1:250000 scale geo-rectified space-maps, based on seasonally standardised Landsat TM satellite imagery captured primarily during 1994-95 (Thompson, 1996; Fairbanks *et al.*, 2000). For the purpose of the present study the 31 land-cover classes were reclassified into three categories, namely natural vegetation, modified vegetation and transformed (Table 1; Figure 2) (based on Wessels *et al.*, 2000 (see Addendum II)). Natural vegetation included all untransformed vegetation, e.g. forest, woodland, thicket and grassland. The modified vegetation category was dominated by degraded classes of land-cover. These areas have a very low vegetation cover in comparison with the surrounding natural vegetation cover and were typically associated rural population centres and subsistence level farming, where fuel-wood removal, over-grazing and subsequent soil erosion were excessive (Thompson, 1996).

The transformed category consisted of areas where the structure and species composition were completely or almost completely altered (Poore, 1978) and includes cultivated, afforested or urbanised areas, as well as mines and quarries. The average thematic mapping accuracy for the province was 73%, with much of the error being attributed to misclassification in bushland-woodland transition zones, not in identifying human land-use impacts (Fairbanks & Thompson, 1996; Fairbanks *et al.*, 2000).

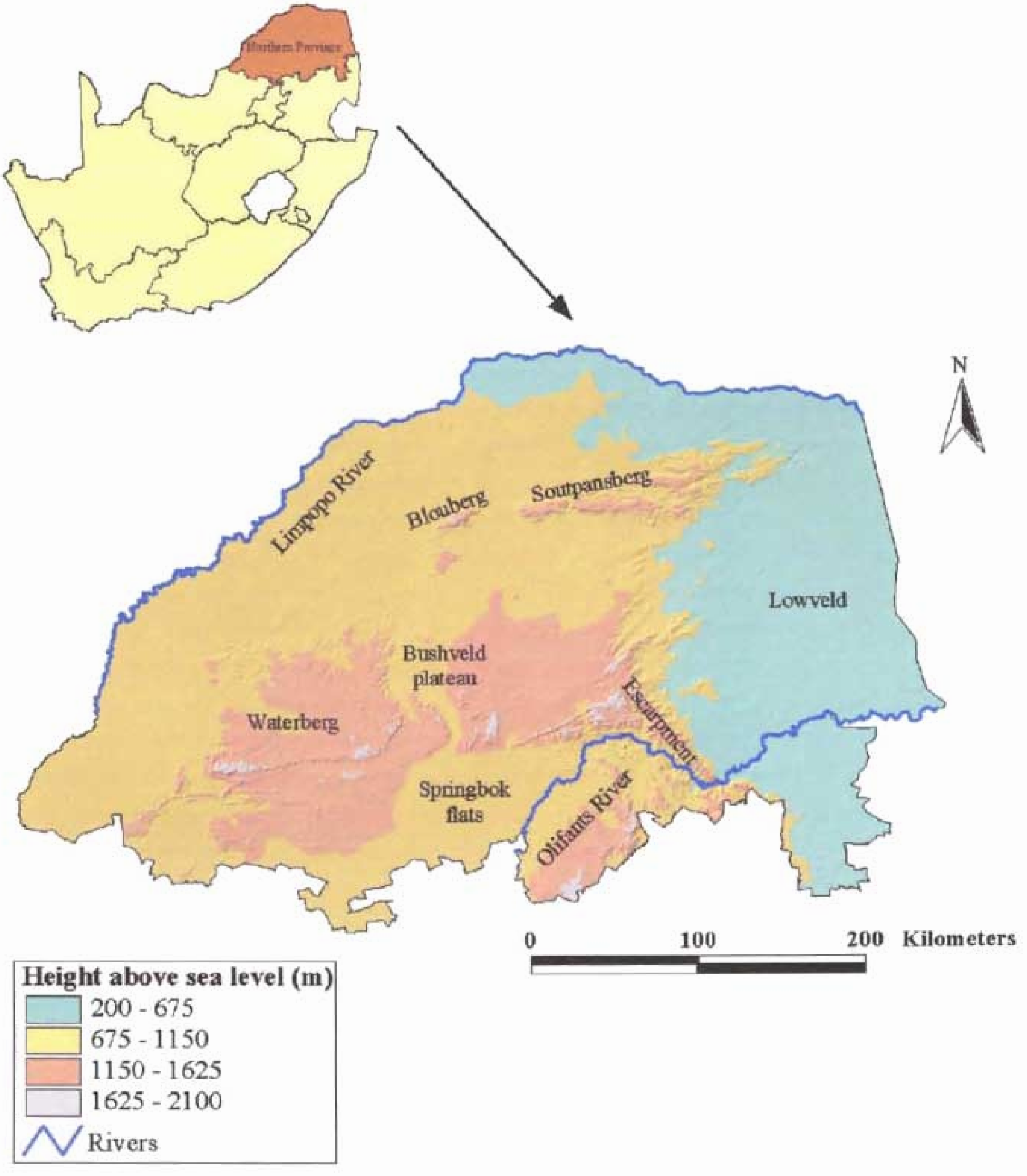


**Figure 1:** Map of the Northern Province study region with major geographical features.

**Table 1:** Land-cover classes reclassified into transformation categories (after Wessels *et al.*, 2000 (see Addendum II)).

| Transformation category | % area | Land-cover class |
|---|---|---|
| Natural vegetation | 73.4% | Wetlands, grassland, shrubland, bushland, thicket, woodland, forest |
| Modified vegetation | 10.1% | Degraded land (9.9%), erosion scars (0.1%), waterbodies (0.1%) |
| Transformed | 16.5% | Cultivated lands (14%), urban/built-up areas (1.6%), mines and quarries (0.01%), forestry plantations (0.8%) |

**Table 2:** Codes and definitions of explanatory variables used in canonical correspondence analysis.

| **Code** | **Definition** |
|---|---|
| Topography | |
| DEMMEAN | Elevation (m) |
| DEMSTD | Elevation heterogeneity (std. Deviation) |
| Climate | |
| GDMEAN | Number of days per annum on which sufficient water is available for plant growth |
| MAP | Mean annual precipitation (mm) |
| GTMEAN | Annual mean of the monthly mean temperature (°C) weighted by the monthly GD |
| NGTMEAN | Mean temperature (°C) during negative water balance |
| MAT | Mean annual temperature (°C) |
| MAXMNTHMN | Mean temperature of the hottest month, usually January (°C) |
| MINMNTHMN | Mean temperature of the coldest month, usually July (°C) |
| EVANNMN | Total annual pan evapotranspiration (mm) |
| PSEAS_MN | Precipitation seasonality from the difference between the January and July means |
| TSEAS_MN | Temperature seasonality from the difference between the January and July means |
| MXSEAS_MN | Maximum temperature seasonality from the difference between January and July |

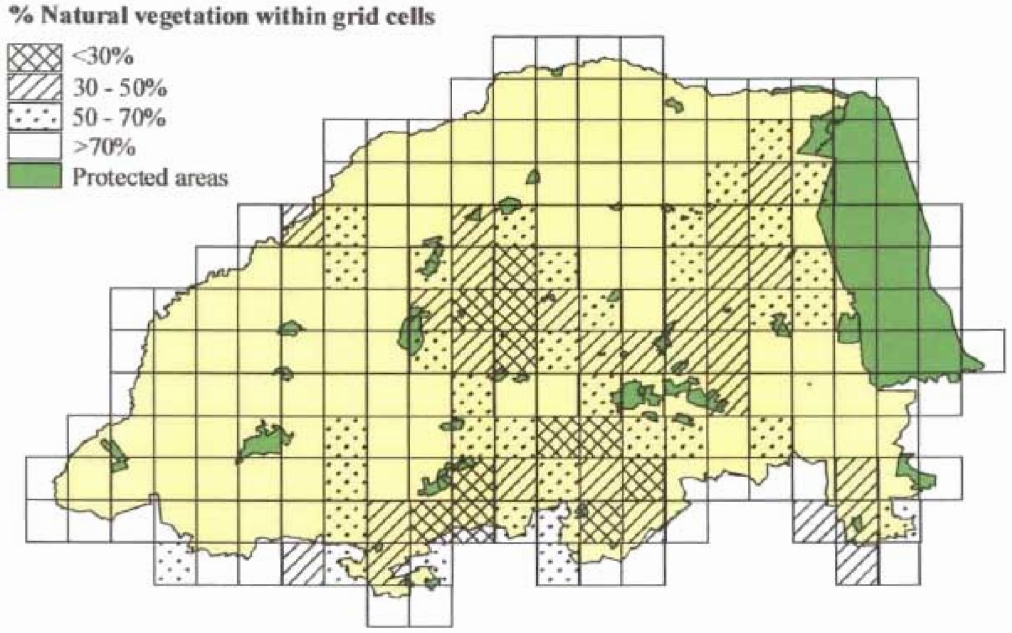


**Figure 2:** Map of the study region illustrating the percentage of natural vegetation remaining in each grid cell. National and provincial protected areas are also indicated.

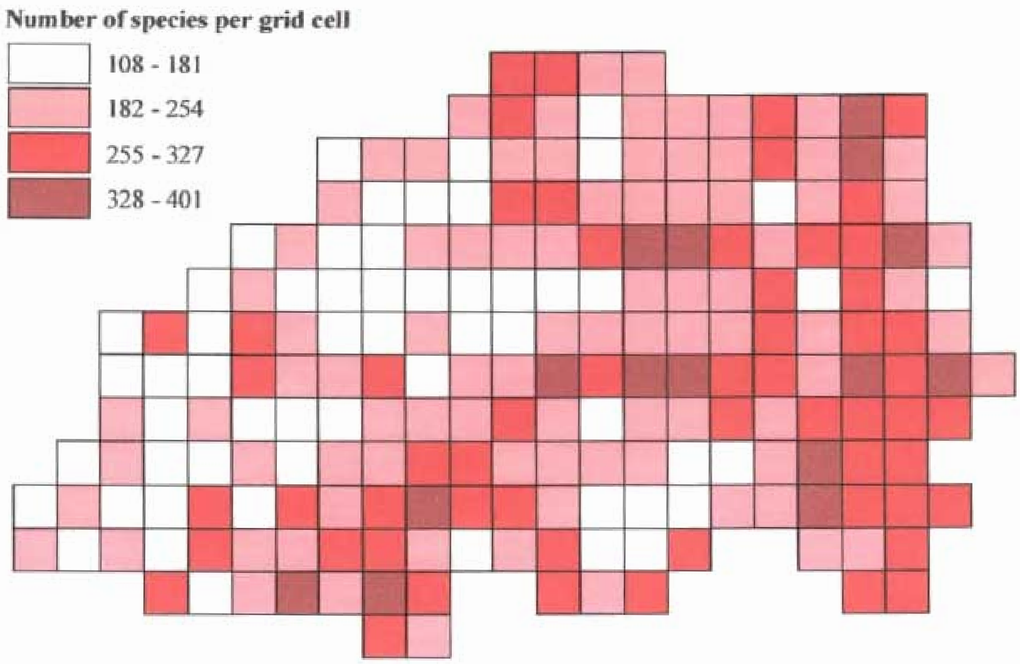


**Figure 3:** Avian species richness within each grid cell of the Northern Province

*GIS analysis*

The environmental and land-cover data were overlaid with the 15' x 15' grid. Each grid cell had an aggregated mean statistic recorded of the environmental and topographical features found within that grid cell, with the addition of the standard deviation of the elevation also being recorded. The extent of land-cover classes, as well as national and provincial protected areas within each grid cell were calculated using ArcInfo (Figure 2) (Albers equal area projection).

*Ordination analysis*

Our primary analytical tool was canonical correspondence analysis (CCA; ter Braak & Prentice, 1988), a direct gradient analysis method used widely in community ecology (Palmer, 1993), and detrended correspondence analysis (DCA), an indirect gradient analysis method (Gauch, 1982). The program CANOCO, version 4.0 (ter Braak & Smilauer, 1998) was used to conduct all gradient analyses. DCA was used to determine the dominant avian species communities within the Northern Province. Environmental data (e.g. the 12 environmental parameters found under topography and climate in Table 2) were entered with the species data using stepwise CCA to investigate which environmental variables explained the patterns in observed avian diversity (ter Braak & Smilauer, 1998).

*Spatial autocorrelation analysis*

In order to combine this information on species patterns and the environmental gradients responsible for those patterns into practical conservation planning techniques, we employed spatial autocorrelation analyses. This form of analysis identifies areas with high levels of species and associated environmental gradient turnover. Using Moran's *I* analysis based on the information gained from the previous ordination analyses we identified local spatial clusters of integrated species composition and their associated environmental gradients (Fairbanks *et al.*, In Press). A grid cell with a high positive Moran *I* value is highly autocorrelated, or similar, to neighbouring grid cells in terms of the avian species it contains i.e. its species assemblage structure as well as environmental characteristics responsible for the presence of those species. A grid cell with a negative Moran *I* value shows a low level of autocorrelation and is thus very different from surrounding grid cells in terms of its species assemblage composition and the environmental variables associated with those assemblages. Thus those grid cells with low levels of spatial autocorrelation are indicative of areas with high turnover in species diversity as well as the environmental gradients responsible, and should be included into conservation area selection procedures.

*Conservation area selection*

Richness-based complementary algorithms (Kirkpatrick, 1983; Howard *et al.*, 1998; Chapters 2 & 3) were initially run on the bird species distribution data. However, as mentioned previously and as evidenced by the results, species-based conservation area selection does not successfully select areas for the representation of natural processes responsible for generating biodiversity patterns. Nor does it target areas of high beta diversity i.e. areas with a high turnover in feature diversity. Thus, the traditional techniques of complementarity based algorithms were adapted to include steps that select areas important to the representation of alpha species diversity patterns, as well as beta species diversity and their associated environmental gradients or processes.

The spatial analysis results enabled the addition of not only species assemblage structure and turnover into reserve selection procedures, it also allowed for the inclusion of those environmental characteristics responsible for assemblages and assemblage turnover. Moran's $I$ values were used as indicators of the importance of a grid cell in terms of species and environmental turnover with respect to neighbouring grid cells. This was done by employing Moran $I$ values as an indication of the uniqueness of the species assemblages contained in that grid cell, as well as the uniqueness of the underlying environmental factors, with respect to the neighbouring cells (Fairbanks *et al.*, In Press). This then made it possible to include the representation of not only alpha diversity patterns, but also beta diversity patterns and the underlying environmental gradients of these patterns into a conservation area selection procedure. Thus the traditional complementarity-based algorithm was reprogrammed to select areas that were high in complementary species richness and low in spatial autocorrelation.

This was done by categorising Moran $I$ values of each grid cell into four groups: negative autocorrelation, weak positive autocorrelation, moderate positive autocorrelation and strong positive autocorrelation. The algorithm started by selecting grid cells with a low level of spatial autocorrelation (i.e. grid cells in the first category of negative autocorrelation), if there was more than one grid cell within the category then complementary species richness of the grid cells was used to resolve ties. The algorithm then proceeded through all spatial autocorrelation categories until all species were represented at least once. In this way grid cells were selected with a high complementary species richness (high alpha diversity), but also with highly dissimilar species compositions and related environmental characteristics from neighbouring and previously selected grid cells (high beta diversity). This beta diversity (BD) algorithm, therefore, selects a network that not only represents all species in the area, but also the unique species assemblages, heterogeneous areas, transition zones, and environmental gradients, i.e. it samples both biodiversity pattern and process in a representative manner.

To identify a conservation area network that reduces conflict with other land-uses and avoids areas that are largely modified and transformed, the algorithms were then modified to successively exclude from selection grid cells that were more than 10, 20 ... 90% transformed and modified (Lombard *et al.*, 1997; Wessels *et al.*, 2000 (see Addendum II)). In essence, this land-use constrained (LUC) algorithm was initially limited to select only grid cells that contained more than 90% natural vegetation until no new species could be added to the system. After that it proceeded in a step-wise fashion to select grid cells that contained less than 90, 80 ... 10% natural vegetation, until all species were represented. The LUC algorithm was therefore based on a trade-off between the primary objective of avoiding transformed land and a secondary objective of representing all species, including unique species assemblages and heterogeneous transition zones. This land-use constraint option was included into a traditional species richness-based reserve selection algorithm, and was also included into the BD algorithm. This BD algorithm first invoked the LUC before using Moran *I* values to select grid cells. Grid cells where there was a conflict between biodiversity conservation and alternative land-uses could be identified and local scale issues highlighted for further investigation. In all conservation area selection procedures species recorded in grid cells with more than 25% currently protected were assumed to be already represented and were excluded from the reserve selection algorithms.

**Results**

*Ordination analysis*

Geographic patterns of DCA scores are indicated in Figure 4 illustrating the four dominant avian assemblages present in the province. The Lowveld community in the east, the central Bushveld plateau community, the Limpopo river basin community forming the northern and western borders of the province and the Escarpment community at the northern tip of the Drakensberg escarpment all containing unique combinations of species. Eigenvalues and gradient lengths were slightly higher for DCA than for the detrended canonical correspondence analysis (DCCA) for the first two axes (Table 3). This fact together with the strong and significant correlations between the DCA for axis 1 and 2 and the explanatory variables (Table 4) suggested that much of the variation in avian diversity distribution is related to the measured environmental variables. The stepwise CCA reduced the number of significant variables required to explain the variation in species gradients (Table 5). The majority (94%) of the species variation in the Northern Province was accounted for by the explanatory environmental variables of mean growth days, mean minimum monthly temperatures and mean height above sea level.

The CCA results are graphed as a biplot, in which arrow length and direction indicate correlations between explanatory variables and CCA axes, smaller angles between arrows indicate stronger correlations between variables (ter Braak & Smilauer, 1998) (Figure 5). The dominant compositional gradient (axis 1) reflected an altitudinal gradient, which was primarily represented by mean temperature of the coldest month and mean elevation, from the tropical climate of the low lying savanna of the Lowveld to the subtropical savanna of the bushveld plateau. These two variables are moderately correlated with each other, but reflected low inflation factors in the CCA analysis therefore each was able to provide explanation for the species compositional gradients. With minor exceptions, the axis 1 gradient was generally longitudinal from the low lying Lowveld with mild winter temperatures up to the high lying Bushveld plateau areas with colder winter temperatures region (Figure 5).

The second CCA axis was a gradient in growing season moisture stress, from the areas of warm, dry growing seasons at lower elevations to areas of cooler, wet growing seasons (Figure 5).

**Table 3:** Eigenvalues and gradient lengths (1 Standard Deviation) for the first two axes from DCA and CCA of all bird species for the Northern Province.

| | **Axis 1** | | **Axis 2** | |
|---|---|---|---|---|
| | DCA | CCA | DCA | CCA |
| Eigenvalue | 0.13 | 0.11 | 0.07 | 0.08 |
| Gradient length | 1.62 | | 1.42 | |

**Table 4:** Spearman's rank correlations of explanatory factors with axis scores from DCA and intraset correlation coefficients from CCA that included all explanatory variables.

| | Axis 1 | | Axis 2 | |
|---|---|---|---|---|
| | DCA | CCA | DCA | CCA |
| DEMMEAN | -0.4247 | -0.6798 | 0.7551 | 0.6142 |
| DEMSTD | 0.1985 | 0.1187 | 0.6620 | 0.7095 |
| GDMEAN | 0.3270 | 0.231 | 0.6264 | 0.7783 |
| MAP | 0.2855 | 0.2178 | 0.5636 | 0.7313 |
| GTMEAN | 0.2302 | 0.4541 | -0.8602 | -0.7517 |
| NGTMEAN | 0.3445 | 0.5649 | -0.8145 | -0.7029 |
| MAT | 0.3945 | 0.6238 | -0.7992 | -0.6525 |
| MAXMNTHMN | 0.2317 | 0.4421 | -0.8601 | -0.7477 |
| MINMNTHMN | 0.5982 | 0.8155 | -0.6034 | -0.4416 |
| EVANNMN | -0.6531 | -0.581 | -0.3561 | -0.4908 |
| PSEAS_MN | 0.1677 | 0.0671 | 0.6040 | 0.7319 |
| TSEAS_MN | -0.7629 | -0.7482 | -0.1321 | -0.3065 |
| MXSEAS_MN | -0.7907 | -0.8054 | 0.0319 | -0.1538 |

**Table 5:** Inter set correlations of environmental variables for step-wise CCA for first two axes

| Variable | Axis 1 | Axis 2 |
|---|---|---|
| GDMEAN | 0.1765 | -0.6896 |
| MINMNTHMN | 0.8504 | 0.3393 |
| DEMMEAN | -0.7279 | -0.5627 |

*Note*: Sign reflects arbitrary selection of gradient direction by CANOCO. $P < 0.01$

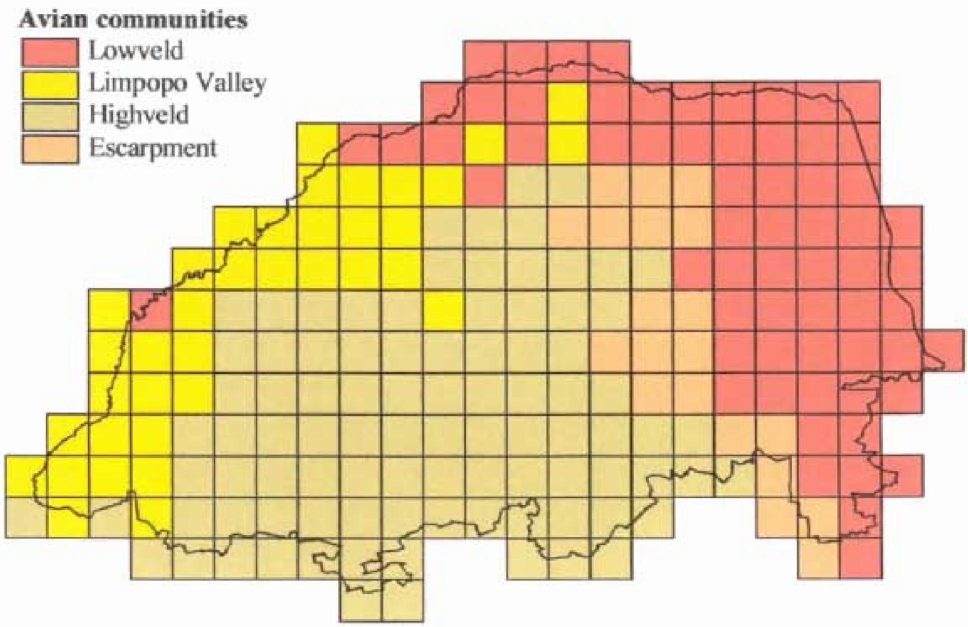


**Figure 4:** Identified avian diversity communities derived from hierarchical classification of first two axes of detrended correspondence analysis results.

Areas of low summer precipitation and high summer temperature include the Limpopo river valley and the low lying Lowveld of the Kruger National Park. The Drakensberg escarpment represents the cool, high summer precipitation, with low evaporation leading to the longer positive water balance.

*Spatial autocorrelation analysis*

This form of analysis was performed on axis 1 and axis 2 of the CCA analysis (Figure 5). The resultant Moran *I* values for both axes were then combined for each grid cell, allowing for the analysis of the species-environment gradient spatial structure. This revealed strong positive autocorrelated clusters of similar species-environment composition in the Lowveld and western Limpopo river valley regions (Figure 6). The Lowveld region is a well-protected region as a result of the Kruger National Park (Figure 4). Negative and weak positive spatial autocorrelated clusters were found in the central Bushveld plateau region representing the Waterberg mountains and the Springbok flats, as well as the northwestern areas representing the Blouberg and Soutpansberg mountain ranges (Figures 1 & 6). These grid cells identify dissimilar species-environment compositions from their immediate neighbours and therefore represent areas of high species turnover of species along the identified environmental gradients. These areas also tended to have poor levels of protection.

*Conservation area selection*

The study region of 214 grid cells included 27 (12.6%) grid cells that were more than 25% protected. These fell mostly within the Lowveld region in the Kruger National Park, and represented 89.73% of the bird species recorded within the province. In order to represent the remaining species, 8.56, 12.83, 15.51 and 18.18% of the province was required by the richness-based, BD, richness-based with LUC and BD with LUC algorithms respectively (Table 6; Figure 7). The traditional richness-based algorithm, although the most efficient in that it represents all species in the least amount of land area possible, because of its selection criteria, concentrates on the areas of high species richness (Figure 3). By selecting these species rich areas mostly in the southern Escarpment and Bushveld plateau regions the richness algorithm tends to avoid areas of negative and weak positive autocorrelation in the northern and western Bushveld plateau (Figure 7). While the BD algorithm, although selecting similar grid cells to the richness algorithm, also selects areas of high turnover in species and environmental variables in these north and northwestern regions of the province (Figure 7). The selection orders of the grid cells by the two algorithms, illustrated in Figures 7a and 7b, demonstrate the different values accorded to each grid cell by the two different approaches. The traditional richness algorithm placing higher precedence on areas with high species richness in the southern regions of the province, while the BD algorithm gives priority to the grid cells in the northern and northwestern regions with high levels of species-environment turnover.

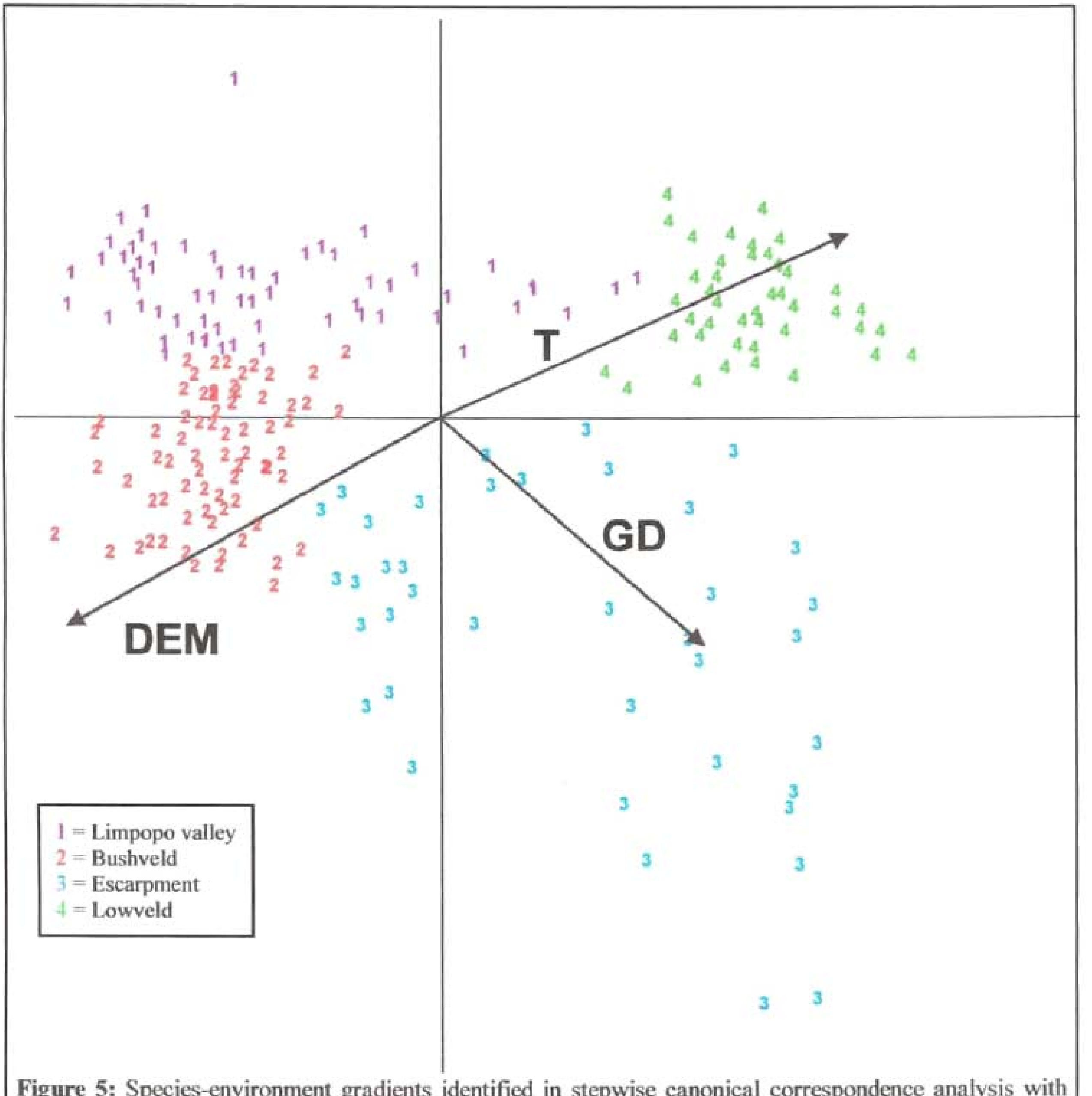


**Figure 5:** Species-environment gradients identified in stepwise canonical correspondence analysis with colours illustrating avian assemblage biogeograhic zones. T = mean monthly temperature of the coldest month, DEM = elevation, GD = growth days (see Table 2)

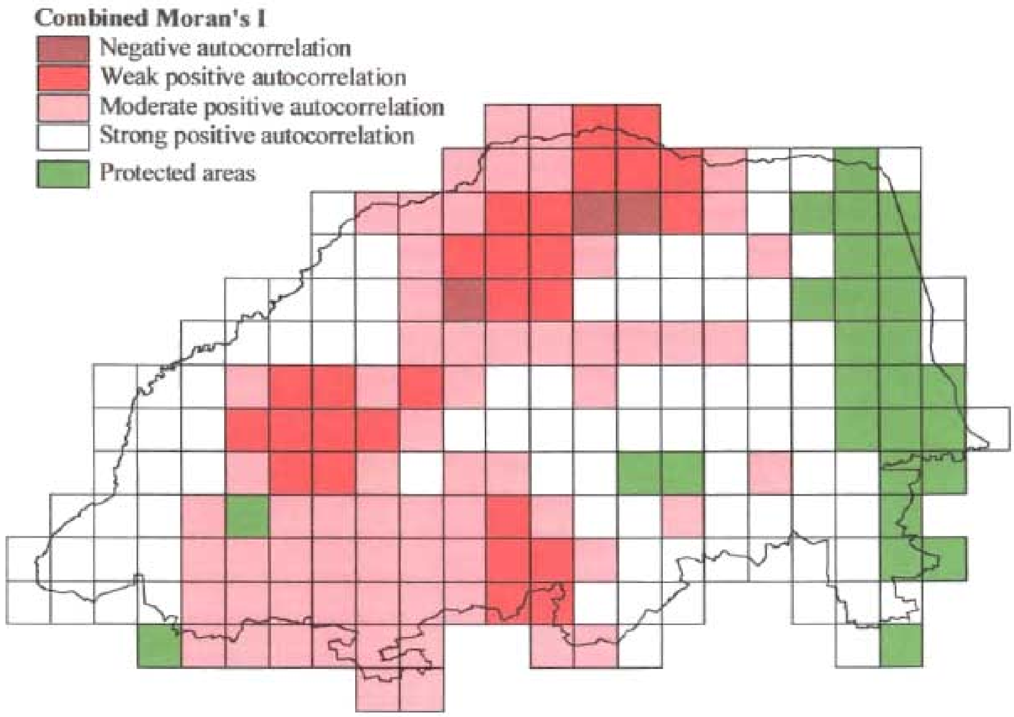


**Figure 6:** Moran's I spatial autocorrelation results for combined Moran's I axes 1 and 2.

**Table 6:** Results of reserve selection algorithms.

| Algorithm type | Number and percentage grid cells selected (with 25% preselection) |
|---|---|
| Richness-based complementary | 16 (8.56%) |
| Beta diversity (BD) | 24 (12.83%) |
| Richness-based complementary with land-use constraint | 29 (15.51%) |
| BD algorithm with land-use constraint | 34 (18.18%) |

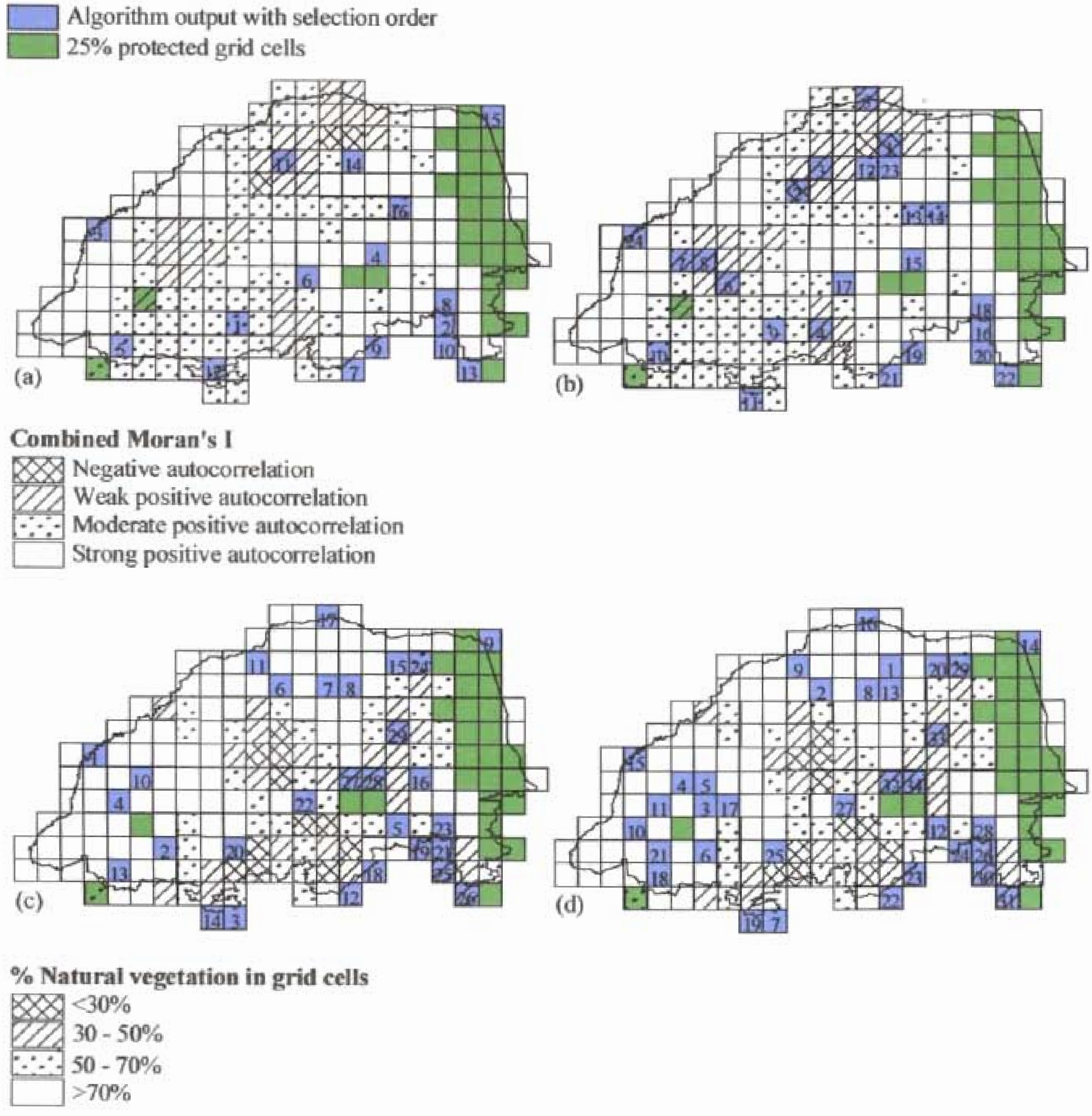


**Figure 7:** Output results for: (a) traditional richness-based complementary algorithm, (b) beta diversity algorithm, (c) traditional richness-based complementary algorithm with land-use constraint, and (d) beta diversity algorithm with land-use constraint. Illustrating degree of spatial autocorrelation within selected conservation area networks (a, b), and percentage of natural vegetation remaining in selected conservation area networks (c, d). Numbers in grid cells indicate selection order of grid cells by algorithms.

Once the land-use constraint is included, the level of efficiency decreases, but the amount of natural vegetation within the conservation area networks increases from 71.3 to 78.92% for the richness-based algorithm and from 74.53 to 80.7% for the BD algorithm (Figures 7 & 8). Although the difference may not seem large, the selection orders of the grid cells within the various networks demonstrate the different values accorded to the grid cells once the land-use constraint is included (Figure 7). The land-use constraint placing higher values on grid cells with low levels of land transformation in the northern and northwestern regions (Figures 7c & 7d)). There is therefore a trade off between land-use efficiency and the representation of species compositional and environmental turnover, as well as a trade off between this efficiency and the avoidance of largely modified and transformed areas.

**Discussion**

Traditional complementary approaches to conservation focus primarily on maximising the conservation of contemporary alpha diversity patterns using measures of species or feature richness (Pressey *et al.*, 1993, Margules & Pressey 2000). But as evidenced by the results of the present study, the use of traditional principles such as complementarity, flexibility and irreplaceability are not sufficient. Although they successfully represent existing biodiversity patterns, they do not guarantee the long-term maintenance of these patterns through conservation of governing natural processes, feature diversity turnover and the reduction of human driven threats on these patterns (Cowling *et al.*, 1999; Balmford *et al.*, 1998; Rodrigues *et al.*, 2000).

*Maintenance of natural processes*

Species, although an important component of the biodiversity hierarchy and a popular focus of many conservation efforts, should not be the only representation goal of conservation area selection (Faith & Walker, 1996a; Maddock & du Plessis, 1999; Noss, 1990; Pressey, 1994a; Wessels *et al.*, 1999). By focussing on only the species level many avian species communities and species-environment gradients important to avian diversity in the Northern Province are potentially ignored or left underrepresented. This underrepresentation results in an inefficient representation of overall regional biodiversity pattern and process.

Existing protected areas within the study area are concentrated mostly within the Lowveld region, most of which is made up by the Kruger National Park. This leaves the other avian communities largely unrepresented. The traditional complementarity based algorithm, does little to correct this representation bias selecting additional grid cells in the relatively homogenous southern Bushveld plateau regions, leaving the northern Bushveld plateau, Escarpment and Limpopo valley avian communities largely unprotected. This is due mostly to the fact that the Lowveld and southern Bushveld plateau regions are highly species rich areas (Figure 3) containing over 90% of the avian species recorded within the province.

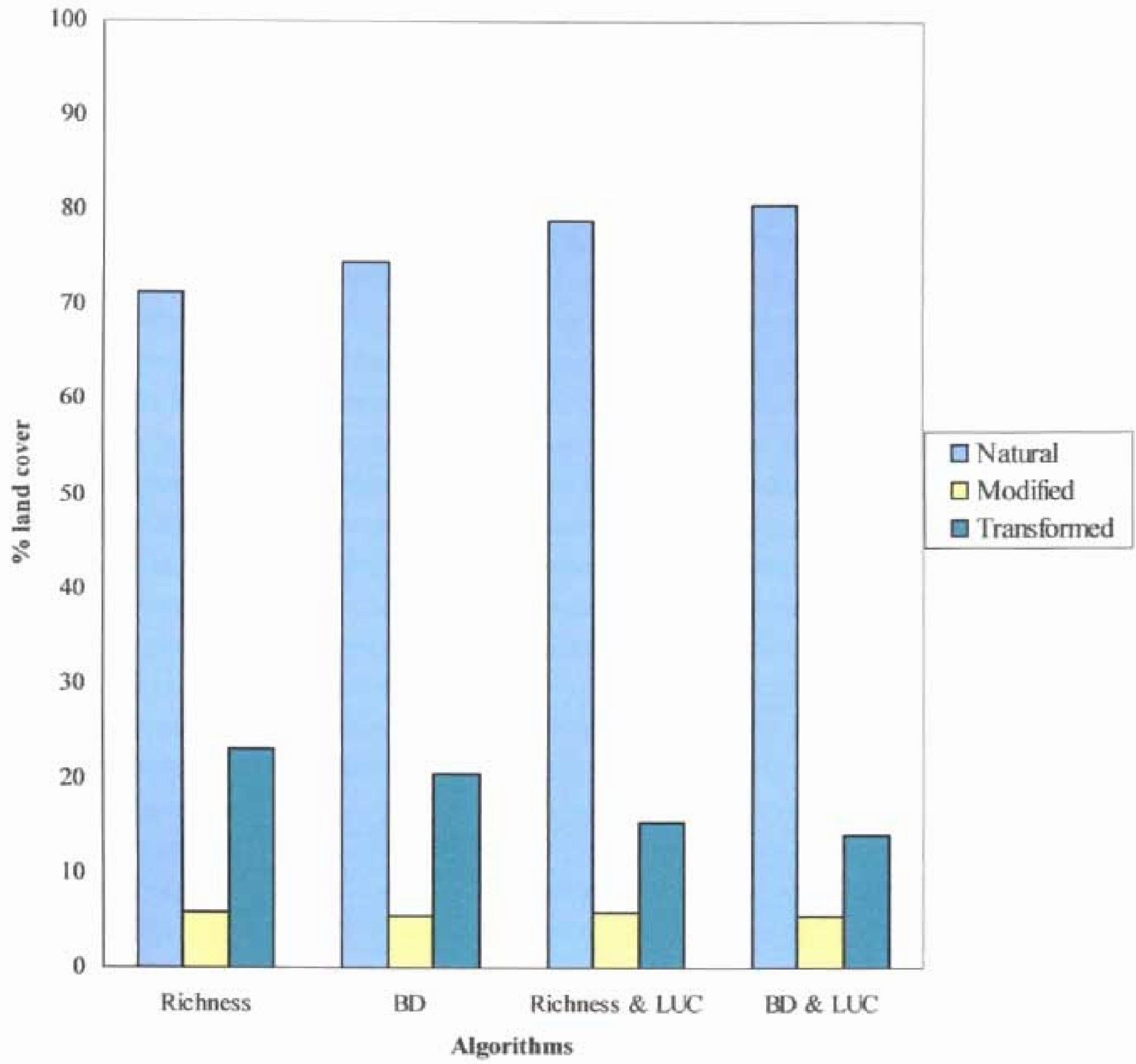


**Figure 8:** Graph of percentage land-cover categories included in conservation area networks selected by richness-based complementary algorithm and beta diversity (BD) algorithm (with and without land-use constraint (LUC)).

Thus, once these areas are represented, almost all of the avian diversity within the province is represented and from a species representation point of view there is no need for additional grid cells.

This makes the investigation and identification of the species community structure as well as the environmental gradients responsible for controlling and maintaining that structure an essential part of conservation area selection procedures. By attempting to protect not only the biodiversity pattern but also the processes responsible for that pattern, we come closer to guaranteeing the representation as well as the long-term maintenance of regional biodiversity. The grid cells selected by the new BD algorithm, although similar to those selected by the traditional algorithm, differ in that some grid cells fall within the other avian communities, particularly the highly heterogeneous areas in the northwestern and central Bushveld plateau community. These areas fall mostly within the Blouberg, Soutpansberg and Waterberg mountain ranges, as well as the Springbok flats region (Figure 1).

In addition to the underrepresentation of many of the avian species-environment communities by the traditional reserve selection procedures, it is obvious from the CCA analyses that they succeed in representing only one extreme of the CCA species-environment gradients. By focusing on species representation alone the low lying, moist, warm Lowveld region is well represented, but the dryer, higher lying, cooler areas with their unique species assemblages are excluded as well as the areas lying between these extremes. The role of the environment as a generator of species distribution patterns has long been a recognised fact (e.g. Whittaker, 1977; Wiens, 1989; Brown, 1995; Maurer, 1998). Fairbanks *et al.* (1996) presented evidence showing that the end points of species-environment gradients, areas where the climate is more stressful for life (overly cold, hot, or dry) (Jongman *et al.*, 1995), were found to be more strongly affected by climate change and therefore resulted in a possible species composition change. A South African climate change study conducted on invertebrate and vertebrate taxa estimated that 66% of all species found within the Kruger National Park, one extreme of the species-environment gradients, will have a less than 50% chance of being found there after a doubling of $CO_2$ levels (van Jaarsveld *et al.*, 2000). An issue often discussed but rarely applied in conservation biology regards the effect of climate change on current conservation areas and future conservation planning (Peters & Darling, 1985; Balmford *et al.*, 1998; Huntley, 1998).

Therefore, although the BD algorithm is less land-use efficient, requiring four percent more land area, it manages to represent high levels of species richness and avian communities, as well as the identified species-environmental gradients in the final proposed conservation area network. Thereby ensuring the representation of natural processes, and consequently the long-term maintenance of regional biodiversity in the face of less obvious human disturbances (e.g. climate change and altered fire regimes) as well as longer-term evolutionary and ecological processes (Balmford *et al.*, 1998; Cowling *et al.*, 1999).

*Turnover in feature diversity*

Spatial autocorrelation analysis proved to be a valuable tool in the identification of areas of high beta diversity, as opposed to employing simple measures of alpha diversity traditionally used by reserve selection techniques. Moran *I* values for both the altitudinal-temperature species-environment gradient of axis 1 and the water balance species-environment gradient of axis 2 from the CCA analysis (Figure 4) enabled the identification of areas high in beta diversity. These areas highlighted by low Moran *I* values contained very different species assemblages from their neighbouring grid cells, as well as different environmental variables controlling these assemblages. By focussing on grid cells with low levels of spatial autocorrelation, the BD algorithm identified areas with highly dissimilar species, community, and environmental compositions from neighbouring grid cells in the Waterberg, Blouberg and Soutpansberg mountain ranges as well as the Springbok flats region. These areas contain unique environmental characteristics not found in surrounding areas. The northern mountain ranges of the Blouberg and Soutpansberg are the meeting point of the southern limit of the Zambezi flora and the northern limit of the South African faunal and floral species. The Sprinbok flats is a basin surrounded by the Waterberg and escarpment in the Northern Province and by the Magaliesberg, a mountain range in the Gauteng Province which lies to the south of the study area. Due to its clay substrate the basin has poor drainage and contains wetlands as well as part of the Olifant River in the east and is therefore considered a birding hotspot. Thus these regions identified by the spatial analysis are important areas with unique species assemblages and environmental characters and are sites of high turnover in species diversity along the identified environmental gradients.

The different selection orders (Figure 7) of the richness-based and BD algorithms illustrate the highly dissimilar approaches and values assigned to each grid cell by the two algorithms. The traditional richness method favouring areas of high species richness (Lowveld and southern Bushveld regions) and the BD method placing more importance on areas containing highly dissimilar species and environmental compositions from surrounding areas (northern and central Bushveld plateau). In order to identify conservation area networks that are robust to turnover in feature diversity, the identification of dominant environmental gradients controlling turnover in species composition using ordination techniques like CCA is important for understanding future environmental sensitivities and evolutionary potential (Cowling *et al.*, 1999; Noss, 1996). This spatial autocorrelation method allows for the incorporation of measures of beta diversity into what are traditionally alpha diversity based reserve selection techniques. The results of the present study illustrate the value of the inclusion of areas with high levels of alpha and also beta diversity. The mountain ranges and other regions highlighted by these analyses are areas of high turnover in avian diversity along unique environmental gradients, and under the existing protected area system are left largely unprotected.

*Minimising threats*

Since the early 1900's the area of cultivated land within South Africa has increased from 3 to 8% (Scotney *et al.*, 1988). Despite this increase the Northern Province remains largely untransformed and includes many land-uses considered to be more amenable with biodiversity conservation (Pressey, 1992), e.g. wildlife reserves, game ranching and livestock grazing. Including these largely untransformed areas into potential conservation areas and attempting to avoid conflict with other land-uses entails selecting a larger area. However, minimising land area requirements through efficient complementarity approaches often involves selecting highly transformed areas that may not be able to sustain species or ecological processes over the long-term (Freemark, 1995). These areas are therefore impractical conservation options and should be precluded from conservation area networks (Wessels *et al.*, 2000 (Addendum II)).

The present study does not demonstrate a large difference in the proportion of transformed land within conservation areas (seven percent difference) selected by techniques with or without the land-use constraint. This is probably due to the fact that the Northern Province, with over 70% natural land-cover, does not contain large areas of transformed and degraded land, thus the chances of selecting transformed land during reserve selection is rare. However, the selection order of the grid cells by the various algorithms illustrates how the land-use constraint gives preference to largely untransformed areas in the northern and western regions of the study area. It attempts to avoid the Springbok flats with high levels of commercial dryland cultivation, as well as the central, southwestern and escarpment regions with combinations of commercial and subsistence dryland cultivation and the associated erosion and degradation. The LUC algorithms select grid cells with less than 50% natural vegetation in these transformed and degraded regions only as a last resort if the species they contain are not present in any of the other less transformed grid cells.

This is similar to the finding by Nantel *et al.* (1998) and Wessels *et al.* (2000) (see Addendum II) that it is often the irreplaceable grid cells, containing some of the rarest species, which are more potentially conflicting than others, being closer to human populations, infrastructure and agriculture. This then enables one to investigate the potential conflict between conservation interests and land-uses within these irreplaceable grid cells at a local scale. Appendix 1 contains a list of the species found only within grid cells that are more than 50% transformed and is an example of how local scale potential conflict issues can be highlighted, specifically in the case of threatened and endangered species (Wessels *et al.*, 2000 (see Addendum II)). Fortunately in the case of the Northern Province it would appear that of the seven bird species found only within these transformed and irreplaceable cells in the study region, most do in fact occur elsewhere in southern Africa and often in the rest of Africa. The Ground Woodpecker, a globally threatened species, and the Caspian Tern, listed as rare in the South African Red Data Book are the only species of special conservation interest, but occur either widely in South Africa or else in the rest of the world. The Bluespotted Dove and Grey Waxbill require monitoring, but are not particularly threatened (Harrison *et al.*, 1997).

**Conclusion**

South Africa, ranked as one of the top 25 richest countries in the world in terms of its biodiversity (WCMC, 1992), faces large threats through human land-use and transformation (Macdonald, 1989; Fairbanks *et al.*, 2000). Existing conservation measures required to address and prevent these land-use changes are largely inadequate, with much of the country's biodiversity wealth lying outside of formal protected areas (Rebelo & Siegfried, 1992; Freitag *et al.*, 1998). This inadequacy of existing conservation measures is not unique to South Africa and is in fact a worldwide phenomenon. The techniques required to redress this issue are largely inadequate and seldom implemented. The issues of process and pattern maintenance, turnover and land transformation that we attempted to address proved to be important issues in reserve selection.

The benefits of the current generation of iterative complementarity based reserve selection algorithms are well-cited (Pressey *et al.*, 1993). These techniques do have their value and are an improvement on the largely *ad hoc* methods of the past. However the shortcomings associated with the use of these techniques alone are obvious and are a major contributor to their lack of implementation in conservation and land-use planning. The framework of complementarity analysis makes a large contribution to assessing the selection of important species assemblages for conservation, but extending this tool to capture the underlying processes ensures the maintenance of those assemblages.

The ordination and spatial structure additions to traditional complementarity based algorithms make several contributions to biodiversity conservation. These additions improve our knowledge on important environmental factors responsible for biodiversity patterns. They also identify species and environments that are currently underrepresented or threatened, as well as areas of high species turnover along the associated environmental gradients, areas which are often severely vulnerable to land transformation. By protecting these areas we can allow for possible changes in species turnover due to climate change, something which would not be possible using only species-based conservation area identification.

The benefit of maximising the area of natural habitat within a conservation area network carries with it the cost of higher land area requirements. However the inclusion of a land-use constraint makes for far more feasible conservation options by including largely untransformed areas into conservation networks. The identification of potential conflict within areas that are irreplaceable in terms of biodiversity conservation and also largely transformed is an important component of regional biodiversity assessments (Wessels *et al.*, 2000 (see Addendum II)). This enables the identification of crucial habitats within these areas required for the continued survival of specific species, as well as the investigation of land-use circumstances within the ranges of other important species.

However, it is unlikely that all the areas identified within these analyses can be formally protected. The future of conservation in South Africa, as in many other developing countries, is uncertain. The lack of resources for the expansion of current formal protected areas in an effort to address the inadequate conservation efforts existing, evidenced by the largely biased current representation of the Lowveld region only in the Northern Province, is not the only problem facing South African conservation efforts. Land reform proposals and the redistribution of privately and state owned land to small-scale subsistence farmers could potentially conflict with conservation objectives. But, the development of effective and scientifically sound techniques for the identification of areas important to conservation need not only be limited to the expansion of existing formal protected areas, but also has an important role to play in the identification of conservation areas to be managed in the human matrix outside of formal protected area networks.

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**References**


Balmford, A., Mace, G.M. & Ginsberg, J.R. (1998). The challenges to conservation in a changing world: putting processes on the map. In *Conservation in a changing world.* (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg ), pp 1-28.Cambridge University Press, UK.

Baudry, J. (1993). Landscape dynamics and farming systems: Problems of relating patterns and predicting ecological changes. In *Landscape Ecology and Agroecosystems* (Eds. R.G.H. Bunce, , L. Ryszkowski & M.G. Paoletti), pp 21-40. Lewis Publishers, London.

Bray, J.R. & Curtis, J.T. (1957). An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs* 27: 325-349.

Brown, J.H. (1995). *Macroecology*. University of Chicago Press, Chicago.

Church, R.L., Stoms, D.M. & Davis, F.W. (1996). Reserve selection as a maximal covering location problem. *Biological Conservation* 76: 105-112.

Cowling, R.M., Gibbs Russell, G.E., Hoffman, M.T. & Hilton-Taylor, C. (1989). Patterns of plant species diversity in southern Africa. In *Biotic diversity in southern Africa* (Ed. B.J. Huntley), pp 19-50. Oxford University Press, Cape Town, South Africa.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G. & Ellis, A.G. (1999). From representation to persistence: requirements for a suistainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa. *Diversity & Distributions* 5: 51-71.

Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L., Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Manuela, H. & Sahr, K. (1997). A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80: 83-97.

DEAT. (1997). *White paper on the conservation and sustainable use of South Africa's Biological Diversity*. Department of Environmental Affairs and Tourism, Pretoria.

Dale, V.H., Pearson, S.M., Offerman, H.L. & O'Neill, R.V. (1994). Relating patterns of land-use change to faunal biodiversity in central Amazon. *Conservation Biology* 8: 1027-1036

Di Benedetto, L., Luciani, F., Maugeri, G., Poli Marchese, E. & Razzara, S. (1993). Role of natural vegetation in the agricultural Landscape for biological conservation in Sicily. In *Landscape Ecology and Agroecosystems* (Eds. R.G.H. Bunce, L. Ryszkowski & M.G. Paoletti), pp 131-138. Lewis Publishers, London:

ESRI. (1998). *Arc and Grid Command References*. Environmental Systems Research Institute, Redlands, California.

Fairbanks, D.H.K., McGwire, K.C., Cayocca, K.D., LeNay, J.P. & Estes, J.E. (1996). Sensitivity to climate change of floristic gradients in vegetation communities. *In GIS and Environmental Modelling: Progress and Research Issues* (Eds. M.F. Goodchild, L.T. Steyaert, B.O. Parks, C. Johnston, D. Maidment, M. Crane & S. Glendinning), pp 135-140. GIS World Books, Fort Collins, USA.

Fairbanks, D.H.K., Reyers, B. & van Jaarsveld, A.S. (*In Press)*. Hierarchical species, communities, and ecosystem gradient representation: unrealised potential for retention of avian diversity. *Biological Conservation*.

Fairbanks, D.H.K. & Thompson, M.W. (1996*)*. Assessing land-cover map accuracy for the South African land-cover database. *South African Journal of Science* 92: 465-470.

Fairbanks, D.H.K., Thompson, M.W., Vink, D.E., Newby, T., van den Berg, H.M. & Everard, D.A. (2000*)*. The South African land-cover characteristics database: a synopsis of the landscape. *South African Journal of Science* 96: 69-82

Faith, D.P. (1995*)*. *Biodiversity and regional sustainability analysis*. CSIRO, Division of Wildlife and Ecology, Canberra, Australia.

Faith, D.P. & Walker, P.A. (1996a*)*. Environmental diversity: on the best possible use of surrogate data for assessing the relative biodiversity of sets of areas. *Biodiversity & Conservation* 5: 399-415.

Faith, D.P. & Walker, P.A. (1996b*)*. Integrating conservation and development: incorporating vulnerability into biodiversity-assessment of areas. *Biodiversity & Conservation* 5: 417-429.

Freemark, K. (1995*)*. Assessing effects of agriculture on terrestrial wildlife: developing a hierarchical approach for the US EPA. *Landscape & Urban Planning* 31: 99-115.

Freitag, S., Nicholls, A.O. & van Jaarsveld, A.S. (1998*)*. Dealing with established reserve networks and incomplete distribution data sets in conservation planning. *South African Journal of Science* 94: 79-86.

Gauch, H.G. (1982*)*. *Multivariate analysis in community ecology*. Cambridge University Press, New York.

Harrison, J.A. (1992*)*. The Southern African Bird Atlas Project: five years of growth. *South African Journal of Science* 88: 410-413.

Harrison, J.A., Allan, D.G., Underhill, L.G., Herremans, M., Tree, A.J., Parker, V. & Brown, C.J. (1997*)*. *The Atlas of Southern African Birds*, Vols 1 and 2. BirdLife South Africa, Johannesburg.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W., Baltzer, M., Dickinson, C.J., Lwanga, J.S., Matthews, R.A. & Balmford, A. (1998*)*. Complementarity and the use of indicator groups for reserve selection in Uganda. *Nature* 394: 472-475

Huntley, B.J. (1998*)*. The dynamic response of plants to environmental change and the resulting risks of extinction. In Conservation in a changing world. (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg), pp 69-86. Cambridge University Press, UK.

Jongman, R.H.G., ter Braak, C.J.F. & van Tongeren, O.F.R. (1995*)*. *Data analysis in community and landscape ecology*. Cambridge University Press, Cambridge.

Kirkpatrick, J.B. (1983*)*. An iterative method for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation* 25: 127-134.

Lombard, A.T., Cowling, R.M., Pressey, R.L. & Mustart, P.J. (1997). Reserve selection in a species-rich and fragmented landscape on the agulhas Plain, South Africa. *Conservation Biology* 11: 1101-1116.

Low, A.B. & Rebelo, A. (1996). *Vegetation of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs and Tourism, Pretoria, South Africa.

Macdonald, I.A.W. (1989). Man's role in changing the face of southern Africa. *In Biotic diversity in Southern Africa. Concepts and conservation*. (Ed. B.J. Huntley), pp 51-77. Oxford University Press, Cape Town, South Africa.

Maddock, A. & du Plessis, M.A. (1999). Can species data only be appropriately used to conserve biodiversity. *Biodiversity & Conservation* 8: 603-615.

Margules, C.R., Cresswell, I.D. & Nicholls, A.O. (1994). A scientific basis for establishing networks of protected areas. In *Systematics & Conservation Evaluation*. (Eds. P.L. Forey, C.J. Humphries & R.I. Vane-Wright), pp 327-350. Clarendon Press, Oxford.

Margules, C.R. & Pressey, R.L. (2000). Review article: Systematic conservation planning. *Nature* 405: 243-253.

Maurer, B.A. (1998). *Untangling Ecological Complexity: The Macroscopic Perspective*. The University of Chicago Press, Chicago.

Nantel, P., Bouchard, A., Brouillet, L. & Hay, S. (1998). Selection of areas for protecting rare plants with intergration of land-use conflicts: a case study for the west coast of Newfoundland, Canada. *Biological Conservation* 84: 223-234.

Nicholls, A.O. (1998). Integrating population abundance, dynamics and distribution into broad-scale priority setting. In *Conservation in a changing world*. (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg ), pp 252-272. Cambridge University Press, UK.

Noss, R.F. (1990). Indicators for monitoring biodiversity: a hierarchical approach. *Conservation Biology* 4: 355-364.

Noss, R.F. (1996). Ecosystems as conservation targets. *Trends in Ecology & Evolution* 11: 351.

Palmer, M. (1993). Putting things in better order: the advantages of canonical correspondence analysis. *Ecology* 74: 2215-2230.

Peters, R.L. & Darling, J.D. (1985). The greenhouse effect and nature reserves. *Bioscience* 35: 707-717

Pressey, R.L. (1992). Nature conservation in rangelands: Lessons from research on reserve selection in New South Wales. *Rangelands Journal* 14: 214-226.

Pressey, R.L. (1994a). Land classifications are necessary for conservation planning but what do they tell us about fauna. In *Future of the Fauna of Western New South Wales* (Eds. D. Lunney, S. Hand, P. Reed & D. Butcher), Chapter 3. The Royal Zoological Society of New South Wales, Australia.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993). Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology & Evolution* 8: 124-128.

Pressey, R.L. & Tully, S.L. (1994). The cost of ad hoc reservation: A case study in western New South Wales. *Australian Journal of Ecology* 19: 375-384.

Poore, D. (1978). Ecosystem conservation. In *Sourcebook for a world conservation strategy*: 25 Morges, Switzerland: General assembly paper GA78/10 add 4 IUCN.

Rebelo, A.G. & Siegfried, W.R. (1992). Where should nature reserves be located in the Cape floristic region, South Africa? Models for the spatial configuration of a reserve network aimed at maximizing the protection of floral diversity. *Conservation Biology* .6: 243-252.

Rodrigues, A.S.L., Gregory, R.D. & Gaston, K.J. (2000). Robustness of reserve selection procedures under temporal species turnover. *Proceedings of the Royal Society, London (B)* 267: 49-55.

Rodrigues, A.S.L., Tratt, R., Wheeler, B.D. & Gaston, K.J. (1999). The performance of existing networks of conservation areas in representing biodiversity. *Proceedings of the Royal Society, London (B)* 266: 1453-1460.

SA Surveyor General. (1993). *Digital elevation model for South Africa*. Department of Land Affairs, Division of Surveying and Mapping, Cape Town, South Africa.

Schulze, R.E. (1998). *South African Atlas of Agrohydrology and Climatology Water Research*. Commission report TT82/96, Pretoria, South Africa.

Scott, J.M., Csuti, B., Estes, J.E. & Anderson, H. (1989). Status assessment of biodiversity protection. *Conservation Biology* 3: 85-87.

Scotney, D.M., Botha, A.D.P., Cloete, J.G., Cowan, G., Dreyer, L., Koch, F.G.L., LeRoux, G.H., Little, A.M., Olivier, J. & Saunders, J. (1988). Agricultural areas of southern Africa. In *Long-term data series relating to southern Africa's renewable natural resources*. (Eds. I.A.W. Macdonald & R.J.M. Crawford), South African National Scientific Programmes Report no. 157, pp 316-336. CSIR, Pretoria, South Africa.

Soulé, M.S. (1991). Conservation: Tactics for a Constant Crisis. *Science* 253: 744-750

Taggart, J.B. (1994). Ordination as an aid in determining priorities for plant community protection. *Biological Conservation* 68: 135-141.

ter Braak, C.J.F. & Prentice, I.C. (1988). A theory of gradient analysis. *Advances in Ecological Research* 18: 272-327.

ter Braak, C.J.F. & Smilauer, P. (1998). *CANOCO Reference Manual and User's Guide to Canoco for Windows*: Software for Canonical Community Ordination (version 4). Microcomputer Power, Ithaca, New York

Thompson, M. (1996). The standard land-cover classification scheme for remote-sensing application in South Africa. *South African Journal of Science* 92: 34-42.

van Jaarsveld, A.S., Chown, S.L., Erasmus, B.F.N., Kshatriya, M. & Wessels, K.J. (2000). *Vulnerability and adaptation assessment of South African animal taxa to climate change*. Report to the Department of Environmental Affairs and Tourism, Pretoria, South Africa

Virolainen, K.M., Virola, T., Suhonen, J., Kuitunen, M., Lammi, A. & Siikamaki, P. (1999). Selecting networks of nature reserves: methods do affect the long-term outcome. *Proceedings of the Royal Society, London (B)* 266: 1141-1146.

WCMC. (1992). *Development of a National Biodiversity Index: a discussion paper prepared by World Conservation Monitoring Centre*. Report on the WCMC, 15 September, 1992.

Wessels, K.J., Freitag, S. & van Jaarsveld, A.S. (1999). The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological Conservation* 89: 21-38.

Wessels, K.J., Reyers, B., Kruger, M., van Jaarsveld, A.S. (2000). Incorporating land-cover information into regional biodiversity assessments in South Africa. *Animal Conservation* 3: 67-79.

Whittaker, R.H. (1972). Evolution and the measurement of species diversity. *Taxon* 21: 213-251.

Whittaker, R.H. (1977). Evolution of species diversity in land communities. *Evolutionary Biology* 10: 1-67.

Whittaker, R.H. & Niering, W.A. (1965). Vegetation of the Santa Catalina Mountains, Arizona II, A gradient analysis of the south slope. *Ecology* 46: 429-452.

Wickham, J.D., Wu, J. & Bradford, D.F. (1997). A conceptual framework for selecting and analyzing stressor data to study species richness at large spatial scales. *Environmental Management* 21: 247-257.

Williams, P.H. (1998). Key-sites for conservation area-selection methods for biodiversity. In *Conservation in a changing world*. (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg ), pp 211-250.Cambridge University Press, UK.

Wiens, J.A. (1989). *The Ecology of Bird Communities*. Volumes 1 and 2, Cambridge Studies in Ecology. Cambridge University Press, Cambridge

**Appendix 1:** Range and conservation status of avian species restricted to grid cells with less than 50% natural vegetation.

| Common name | Species name | Grid cell % natural land-cover | Conservation status | Range |
|---|---|---|---|---|
| Bluespotted Dove | *Turtur afer* | 2330AB (*30-40%*); 2330CC (*30-40%*) | Locally common. Population monitoring required | Restricted range in eastern Zimbabwe, adjacent Mozambique and northeastern South Africa. |
| Caspian Tern | *Hydroprogne caspia* | 2330AD (*30-40%*) | Rare in the South African Red Data Book | Occurs throughout the holoarctic, Australasian, Oriental and Afrotropical regions |
| Goldenrumped Tinker Barbet | *Pogoniulus bilineatus* | 2531AA (*40-50%*) | Common in forest habitat. Not threatened | Widespread African species. Occurs in the eastern lowlands of South Africa, lowland forest of eastern Zimbabwe & Zimbabwe highlands, as well as Mozambique. |
| Grey Wagtail | *Motacilla cinerea* | 2329DD (*40-50%*) | Regular occurrence in southern Africa. | Non-breeding African palearctic migrant. Common in Africa as far south as northern Tanzania. |
| Grey Waxbill | *Estrilda perreini* | 2230DA (*50-60%*) | Not particularly threatened but worth monitoring in southern Africa. | Occurs in Gabon, Angola and Zaire eastwards to Tanzania and Mozambique, and southwards to South Africa. |
| Ground Woodpecker | *Geocolaptes olivaceus* | 2430DD (*40-50%*) | Globally near threatened. Wide range | South African endemic |
| Southern Tchagra | *Tchagra tchagra* | 2430DD (*40-50%*) | Fairly common in its restricted range. Not of particular conservation concern. | Southern African endemic |

Harrison *et al.*, (1997)

# CHAPTER 6

## Incorporating potential land-use threats into regional biodiversity evaluation and conservation area prioritisation

## Introduction

Humans have transformed almost half of the world's ice-free land surface area into agricultural and urban systems. Through this transformation humankind dominates (directly or indirectly) almost one third of the land's net primary productivity and uses 54% of the available freshwater (Vitousek *et al.*, 1997; Chapin *et al.*, 2000). In the absence of policy and behavioural changes, anthropogenic land-use impacts are said to become the largest threat facing terrestrial global biodiversity by the year 2100, particularly within the tropics (Chapin *et al.*, 2000; Sala *et al.*, 2000). Although there are several other drivers of biodiversity change, including changes in atmospheric $CO_2$ levels, nitrogen deposition and acid rain, land-use changes will have the largest impact mostly due to their destructive effects on habitat availability and consequent species extinctions (Sala *et al.*, 2000). Not only will these changes have implications for ecosystems, altering their processes and resilience to environmental change, but will also have important consequences for humankind due to increasingly threatened ecosystem services and products (Kunin & Lawton, 1996; McCann, 2000).

The establishment of conservation areas in which features of biodiversity are separated from processes that threaten their persistence in the wild, is one of the widely used approaches for addressing these threats (Pressey, 1998; Margules & Pressey, 2000). Shortcomings of existing conservation areas, the need for systematic conservation area selection procedures, as well as the need for these procedures to minimise threats facing regional biodiversity within selected areas has been widely researched (for review see Balmford *et al.*, 1998; Williams, 1998; Margules & Pressey, 2000) and discussed in Chapters 1 and 5, as well as Addendum II. The study in Addendum II, conducted in the three northern provinces of South Africa, focused on the need to identify and minimise current land-use threats facing biodiversity. The incorporation of measures of threat into conservation area selection is essential and has important implications for land-use planning, enabling the identification and therefore avoidance of areas of threat to biodiversity and highlighting areas where there may be conflicts between conservation and land-use development issues (Lombard *et al.*, 1997; Nantel *et al.*, 1998, Williams, 1998). These analyses in Addendum II illustrated how human land-use impacts increase the costs of achieving a representative conservation area network, decrease the flexibility of conservation options and in many cases actually conflict with areas irreplaceable to biodiversity conservation.

It is however, also important to remember that human land-use impacts are not static and will continuously evolve and spread as populations and their land-use and resource needs expand. This will subsequently further increase costs, decrease conservation options and increase the amount of conflict between the various forms of land-use and conservation. It is therefore essential that existing natural areas with high potential to become transformed by other land-uses be identified at as early a stage as possible in order to identify areas where there may be future conflict between these potential developments and existing biodiversity. These alternative forms of land-use include agriculture, forestry, mining, and urbanisation, as well as land degradation through overgrazing, fuel wood harvesting and alien plant invasions (Fairbanks *et al.*, 2000). There is thus a need for a conservation area selection technique which avoids areas that are currently largely transformed and also identifies areas crucial to biodiversity conservation requiring management because of high levels of transformation (Addendum II). In addition to this a technique that also identifies untransformed areas that are suitable for development will hopefully help to guarantee persistence of regional biodiversity (Pressey *et al.*, 1996; Williams, 1998).

These untransformed areas identified as highly suitable for alternative land-uses can then, applying the principle of flexibility (Pressey *et al.*, 1993), be avoided by conservation planners and used for development. If however these areas are irreplaceable due to rare or endemic biodiversity features (Pressey *et al.*, 1994; Ferrier *et al.*, 2000) they can be targeted as conservation priorities due to high biodiversity and threat values. A better understanding of the current and future patterns of threats (especially land-use threats) facing biodiversity will allow for more effective trade-offs between biodiversity conservation and development opportunities (Faith, 1995; Faith & Walker, 1996), as well as a more efficient allocation of limited conservation resources for areas most at risk (Margules & Pressey, 2000).

Pressey (1997) highlights the fact that many of the existing conservation area selection techniques say nothing about the relative need of areas selected for protection. Funding and resource shortages dictate that although a large number of areas may be identified as important to the representation of biodiversity, only a small number of them can be protected in the near future. As Cowling *et al.* (1999) point out, in order to maximise the retention of biodiversity features within a region, one must minimise the extent to which the original representation goals are compromised by habitat loss while the conservation area network is developing (a process that can take decades). It is therefore crucial to identify areas of high conservation value or urgency within this selected set of areas. These are areas with a high biodiversity, or irreplaceability value, as well as a high threat or vulnerability value (Faith & Walker, 1996; Pressey *et al.*, 1996; Pressey, 1997; Pressey, 1998; Cowling *et al.*, 1999).

Much work has been done on measuring biodiversity values of areas, and includes species richness, endemism and rarity of areas as well as complementary species richness discussed in Chapters 2 and 3 (Williams *et al.*, 1996; Williams, 1998; van Jaarsveld *et al.*, 1998b). Also included as a measure of biodiversity value are measures of irreplaceablity (Pressey *et al.*, 1994; Ferrier *et al.*, 2000), which illustrate how crucial a site is for achieving representation goals within a region due to its biodiversity feature content. Other measures of biodiversity value focus less on the biodiversity pattern of an area and more on the biodiversity processes within the region (Balmford *et al.*, 1998; Pressey, 1998). These techniques focus on the spatial surrogates for biodiversity processes and include measures of higher levels of the biodiversity hierarchy (Pressey, 1994; Noss, 1996; Maddock & du Plessis, 1999), environmental gradients (Noss, 1996; Cowling *et al.*, 1999) and measures of spatial and temporal turnover (Chapter 5; Rodrigues *et al.*, 2000). However, there is a large need for work on the inclusion of threat or vulnerability values of areas into conservation areas selection. (Chapter 5; Addendum II; Faith & Walker, 1996; Pressey *et al.*, 1996; Williams, 1998).

The present study therefore aims to address several shortcomings with existing conservation area selection techniques in an effort to incorporate current and potential threat values into conservation planning in the Northern Province of South Africa. First areas with high biodiversity value will be identified using all techniques mentioned previously. Second, areas within the province suitable for alternate land-uses will also be determined. The identified areas of high biodiversity value will then be investigated as to their land-use threats from existing and potential land-uses. Finally, using the biodiversity data available for the province, a conservation area network will be identified which avoids areas currently transformed and degraded while representing all known regional biodiversity. The areas selected will then be investigated in terms of their current and potential land-use threats in an attempt to prioritise areas for immediate conservation action.

It is important to note that the current study focuses only on the forms of land-use deemed important and likely threats to biodiversity within the Northern Province. These land-uses include cultivation (both rain-fed and irrigated), afforestation of various species of *Eucalyptus* and *Pinus*, as well as *Acacia mearnsii* (black wattle), and mining and quarrying. Although several other forms of land-use may also have impacts on biodiversity, these future forms of land-use are not taken into account. Livestock grazing may result in structural as well as compositional changes to vegetation, but under controlled conditions, does not usually result in major land-cover transformation or alteration in ecological function, and is considered to be more amenable with biodiversity conservation (Pressey, 1992; Mishra & Rawat, 1998; Allsopp, 1999). In addition to this the impacts of grazing on biodiversity are difficult to quantify and have not been fully investigated within the study region. Although areas of land-cover degradation are often indicative of overgrazing (Thompson, 1996; Newby & Wessels, 1997) and can be used as a current indication of threats to biodiversity, future patterns are difficult to predict. Therefore areas not suitable to the main land-uses of forestry, cultivation or mining were assumed to remain natural and unimpacted. Although urbanisation has direct effects on biodiversity, due to limited urban development in the province only current levels of urbanisation were considered in the determination of threats to regional biodiversity (Rottenborn, 1999).

Road networks, however, were evaluated as to the impacts of this form of infrastructure development on biodiversity. Although road networks occupy small areas, the ecological effects that roads have on regional biodiversity extend far beyond the edges of the road itself (Reijnen *et al.*, 1995; Forman & Alexander, 1998; Forman, 2000). Road networks affect landscapes and biodiversity in seven general ways: (1) increased mortality from road construction; (2) increased mortality from vehicle collisions; (3) animal behaviour modification; (4) alteration of the physical environment; (5) alteration of the chemical environment; (6) spread of exotic species, and (7) increased alteration and use of habitats by humans (from Trombulak & Frissell, 2000). As illustrated in Addendum I, road-effect zones can be used to provide an estimate of the potential threat to regional biodiversity through changing land-uses and increased future human impacts.

## Methods

### *Study area*

The Northern Province lies at an average 880m above sea level, with higher lying areas reaching just over 2000m in the escarpment and low lying regions in the lowveld of the Kruger National Park falling below 300m (see Figure 1 in Chapter 1). This varied elevation results in different rainfall regimes within the region. The province is largely a low summer rainfall area with an average rainfall of 500mm per annum. However there are areas within the higher lying escarpment which receive in excess of 1200mm per year, while other areas in the far north of the province receive less than 300 mm (Schultze, 1998). The temperatures recorded in the province are also variable, with an average temperature of 19°C falling to 16°C in the southern and central high lying highveld region, and climbing to 23°C in the lowlying subtropical lowveld (see Figure 1 in Chapter 1).

In a similar fashion to most of South Africa, the Northern Province has large tracts of untransformed land (Figure 1) (see Figure 4 in Chapter 1; Addendum I). There are few urban centres within the province (see Figure 3 in Chapter 1), therefore urbanisation and industrialisation do not pose a large threat within the province, accounting for only 1.6% of the area. Other forms of infrastructure occupy small areas of the province, e.g. the road network takes up less than 0.02% of the area. The province includes extensive areas of arable land and as a result 14% of the province has been transformed by cultivation. However due to the relatively low rainfall in most parts of the province, dryland (or rain-fed) cultivation at a commercial scale, which makes up two percent of the total cultivation in the province, is limited to the escarpment, mountainous regions and Springbok Flats where it is considered a viable land-use option. In the rest of the province rain-fed agriculture is not possible at a commercial scale and is limited to temporary and subsistence level cultivation, making up 38 and 48% of the total cultivation in the province respectively (see Table 2 in Chapter 1) (Fairbanks *et al.*, 2000).

Other areas under cultivation require irrigation, and because of the scarcity of suitable rivers in the province this form of cultivation is very limited making up three percent of the total cultivation at a commercial level and eight percent at a temporary level (Fairbanks *et al.*, 2000). In a similar fashion, afforestation within the province is limited (0.8% of the province) by the low levels of rainfall to the higher lying and moister escarpment. Despite its mineral wealth mines and quarries occupy only 0.1% of the land area (Wilson & Anhaeusser, 1998).

Thus, although the province is largely untransformed at present, it does contain substantial areas that are suitable for alternate land-uses. Afforestation, cultivation and mining are considered to be major land-uses that threatened biodiversity and all three are viable within the Northern Province.

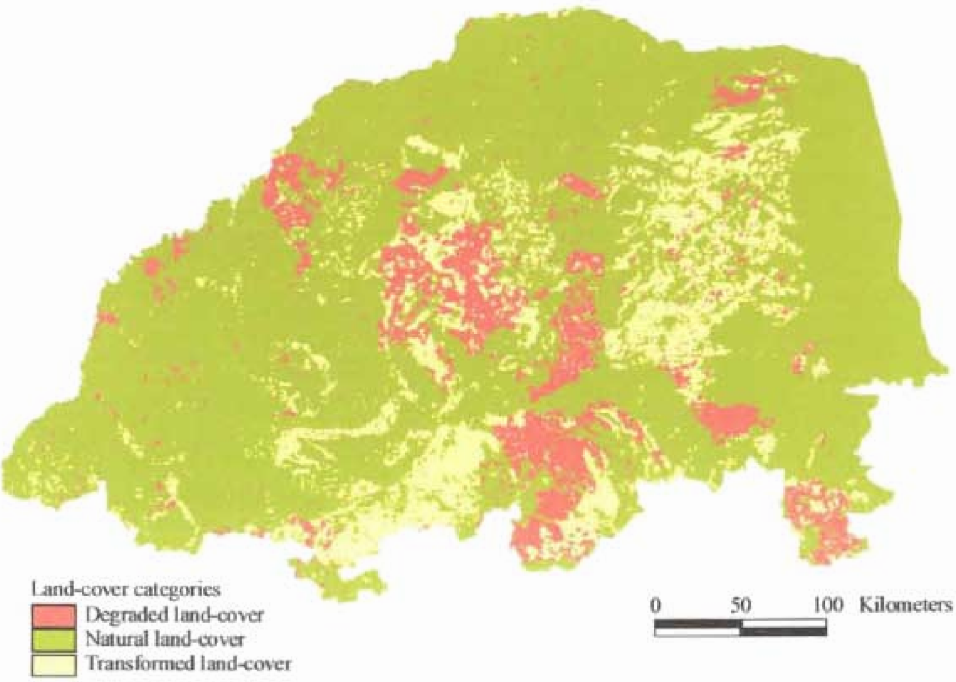


**Figure 1:** Study area of the Northern Province showing areas of land-cover change. Land-cover categories from Chapter 5.

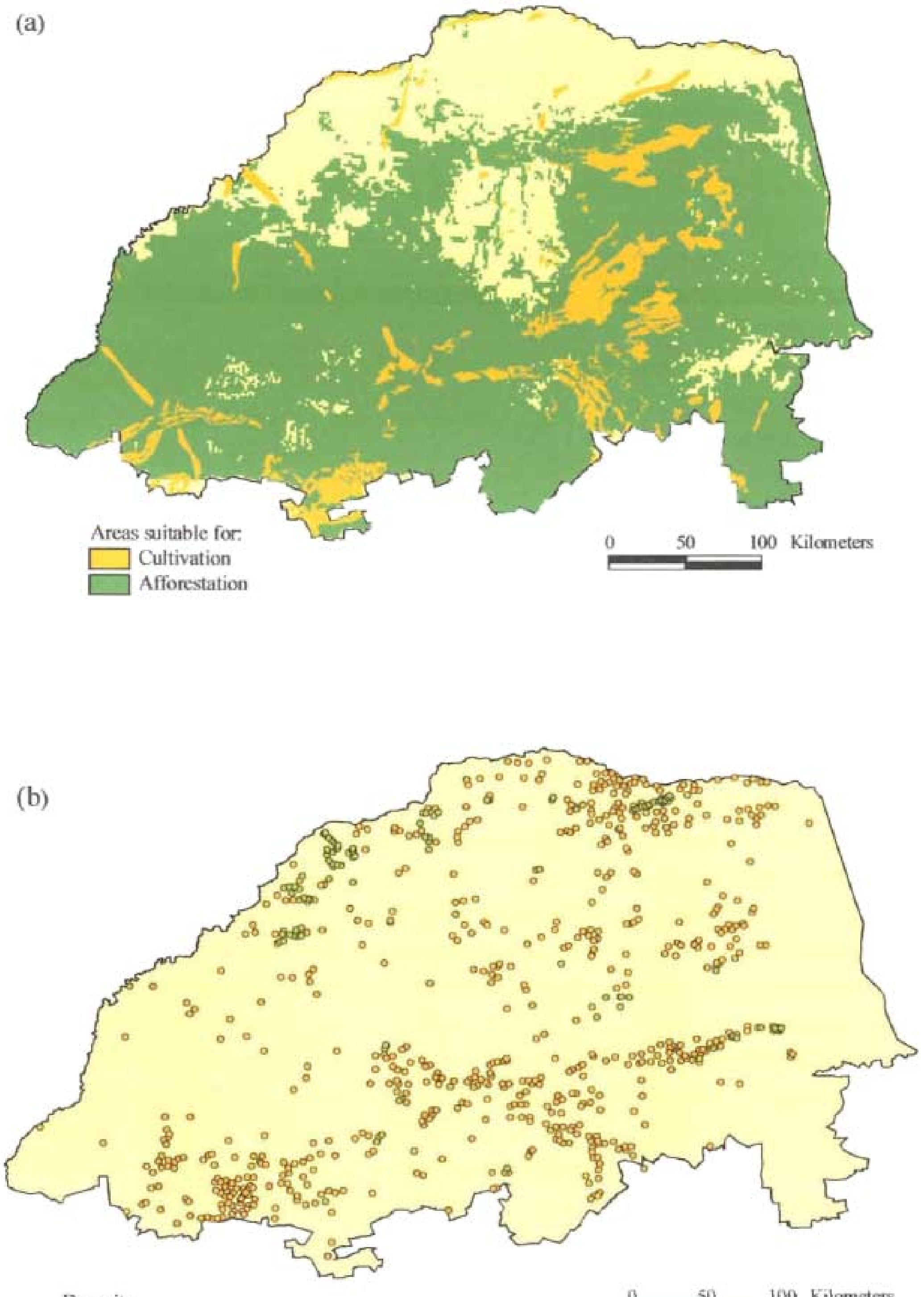


**Figure 2:** Northern Province showing areas of: (a) high suitability for: cultivation (both irrigated and rain-fed) and forestry, and (b) known mineral and dimension stone deposits.

The low rainfall within the province, as well as new water laws within South Africa (DWAF, 1996) limit the potential for further afforestation or dryland cultivation. However there are large areas which would be suitable for cultivation and afforestation (especially of specialised species) within the province (Figure 2a) (Fairbanks, 1997). In addition the province has large mineral fields, as well as unexploited mineral and dimension stone deposits (Figure 2b) which may have serious implications for regional biodiversity. The province includes 15 vegetation types falling largely within the savanna biome, as well as smaller parts of the forest and grassland biome on the escarpment and in the mountain ranges (Low & Rebelo, 1996; see Figure 2 in Chapter 1).

*Databases*

All spatial data used in the study were projected in an Albers equal area projection, based on Clarke 1880 Spheroid with 24° E as the central longitude and –18°S and –32°S as the standard parallels. All analyses were performed in Geographic Information Systems (GIS) using ArcInfo and ArcView (with Spatial Analyst) (ESRI, 1998). Each of the spatial coverages used are described below.

<u>Species distribution data</u>

Distribution data on several vertebrate, invertebrate and plant taxa used to investigate areas of high biodiversity value are available for the study area (see Table 3 in Chapter 1). However due to the taxonomic inconsistencies and survey biases (Freitag & Mansell, 1997; Hull *et al.*, 1998; Muller, 1999; Koch *et al.*, 2000) found within a large majority of these databases it was decided to only include databases on the well studied taxa with sound distribution data and taxonomy. These taxa include the birds, mammals and to a lesser extent the butterflies (Harrison, 1992; Freitag & van Jaarsveld, 1995; Freitag *et al.*, 1998; Muller, 1999). The database included 48803 unique records for 565 bird species, 2062 records for 328 butterfly species and 7040 records for 214 mammal species. In this study most of the analyses focussed on the bird distribution data due to its well-assessed quality and reliability. The other taxa are only employed for the final analysis. These distribution records ranged from point localities to 15' x 15' grid cell records, and were therefore generalised up to the 15' x 15' (~ 700 km$^2$) grid cell resolution for the purposes of this study.

<u>Vegetation data</u>

A GIS layer of the vegetation types defined by Low and Rebelo (1996) was used to determine current and future impacts on these areas within the province. These vegetation types are defined as units with similar vegetation structures, ecological processes and important species (Low & Rebelo, 1996). This layer also serves as a broad-scale surrogate for regional biodiversity (see Chapter 4).

Protected area coverage

A GIS layer of the national and provincial protected areas (DEAT, 1996) was used to determine extent of existing protected areas. These protected areas all fall within IUCN categories I and II. A comprehensive layer of private reserves is currently unavailable.

Land-cover data

This layer utilised the National Land-cover database for South Africa (Thompson, 1996; Fairbanks *et al.*, 2000). This is derived from a series of 1:250000 scale geo-rectified maps, based on seasonally standardised, single date LANDSAT TM satellite imagery captured during the 1994-95 period (Fairbanks & Thompson, 1996; Thompson, 1996; Fairbanks *et al.*, 2000). In a similar fashion to Chapter 5 and Appendices 1 and 2, these 31 land-cover classes were grouped into three categories of natural, degraded and transformed land-cover (Figure 1). The transformed class was then further subdivided into cultivated, forestry, mining and urban areas.

Road-effect zone

The spatial extent of ecological effect of roads, or road-effects, can be used as an ecological indicator that directly represents impacts on biodiversity. The affected distances were estimated in a similar fashion to the one used in Addendum I in a hierarchical manner from Stoms (2000) using estimates of spatial extent of the road-effect zone from reviews mentioned previously, as well as from local studies (Milton & Macdonald, 1988). The road-effect zone of large main roads was assumed to be larger (1km on either side of the road) than that of smaller farm roads (100m) (see Addendum I). This zone was determined from road segments from the South African Surveyor General (1993) 1:500000 scale map series files. These were buffered in a standard geographic information system operation to the distance related to its class. The roads in protected areas were excluded from this analysis as the road-effect in national parks is of limited biodiversity concern.

Suitable areas for afforestation, cultivation and mining

Areas suitable for afforestation by eucalyptus, pine and wattle species, the main species used in the forestry industry within the Northern Province, were evaluated using an afforestation potential land evaluation developed by the CSIR (Fairbanks, 1997). This evaluation uses a GIS modelling approach, based on fuzzy sets logic techniques, using information on climate and soils (Fairbanks, 1997). The suitability for summer rainfall pure types of *Eucalyptus camaldulensis*, *E. nitens*, *E. saligna*, *E. tereticornis*, and *E. urophylla*, as well as *Pinus elliottii*, *P. patula*, *P. taeda* and *Acacia mearnsii* (black wattle) were classified. This evaluation used one minute by one minute grid cell data on several physiologically based climate variables including median annual precipitation, mean annual temperature, mean maxima of the hottest month (January), mean minima of the coldest month (July) and seasonal precipitation patterns. In addition broad soil pattern and soil depth were used from 1:250000 soil types mapped by the Institute for Soil, Climate and Water (ISCW, Agricultural Research Council). Suitabilities of species were then grouped into five potential classes: (see Fairbanks, 1997).

0-20% = Highly unsuitable

20-40% = Unsuitable

40-60% = Low suitability

60-80% = Suitable

80-100% = Optimal

For the purposes of this study only areas with a greater than 60% suitability were considered likely biodiversity threat areas.

Areas suitable for maize, wheat and sorghum cultivation were calculated in two ways. First, potential for rain-fed crop production was mapped from land types (MacVicar, 1974; Land Type Survey Staff, 1986, Schoeman & Scotney, 1987) by the ISCW (ARC). This was based on mean annual rainfall (>550mm), soil depth, soil form, clay percentage and slope at 1:250000 scale (for example see Smith, 1998). Areas with a greater than 60% suitability were then classified as regions suitable for rain-fed or dryland agriculture. Second, potential for irrigated crop production was extracted from Schoeman *et al.* (1986), using landtype information, expert knowledge on irrigation schemes, the availability of water and Landsat MSS data to map extant areas of irrigation.

Data from the Metallogenic Map dataset and SAMINDABA (2000) provided in digital format by the Council for GeoScience was used to estimate the potential impacts of mining and quarrying in the province. Localities of deposits and mines for the top 20 minerals and 5 dimension stone types were buffered with a 1km buffer in order to derive a layer of what the potential ecological impacts of mining and quarrying in the area could be (Table 1). These localities were of varying deposit statuses including:

*Occurrence*: a naturally occurring commodity, usually in outcrop, on which subsurface exploratory work has or has not been carried out or is in progress, and which has not yet been proved to be economically viable or is very unlikely to become viable in future

*Deposit*: an occurrence at which subsurface exploratory work has proved that the quality and quantity of the commodity(ies) are such that exploitation has been, or is currently feasible, or is very likely to become feasible in future. This term automatically applies to all producing mines, past and present.

<u>Potential threat</u>

These layers on potential land-use impacts and road-effect zones will then be combined to provide a layer of all potential land-use threats within the region. This allows for the evaluation of overlap with current land-uses and determination of impact on areas that are as yet untransformed, especially those important to biodiversity conservation.

**Table 1:** Mineral and dimension stone deposits in the Northern Province.

| Mineral or dimension stone type | Number of known deposits |
|---|---|
| Gold | 87 |
| Platinum | 15 |
| Chrome | 21 |
| Titanium | 23 |
| Copper | 119 |
| Lead | 23 |
| Zinc | 5 |
| Nickel | 25 |
| Iron | 65 |
| Vanadium | 26 |
| Manganese | 19 |
| Andalusite | 12 |
| Antimony | 6 |
| Tin | 40 |
| Coal | 6 |
| Fluorspar | 34 |
| Phosphate | 18 |
| Limestone | 23 |
| Magnesite | 38 |
| Vermiculite | 6 |
| Diamond (alluvial) | 9 |
| Diamond (in kimberlite) | 6 |
| Quartzite/Sandstone | 3 |
| Granite/Quartz-porphyry/Syenite | 6 |
| Gabbro/Dolerite/Norite | 6 |
| Shale/Slate/Jaspilite/Schist | 13 |
| Marble | 71 |

*Land-use impacts*

Analyses were performed on these data layers to investigate the current and potential land-use scenarios for the Northern Province. Using the current and potential land-use layers available one could ascertain how much of the area suitable for the various land-uses was actually being used or still remained untransformed as an indication of threats facing the province. Similarly the vegetation types could be assessed as to the amount of natural land remaining within each vegetation type, and the percentage of that land that is suitable for other land-uses. In this way vegetation types could be prioritised for conservation action.

*Biodiversity value*

Because of the high quality of the bird species distribution database it was largely employed in this section of the analysis. Traditional priority conservation areas including richness and rarity hotspots were identified. Due to the fact that there are very few known species limited to the Northern Province, endemic hotspots were not applicable. Areas containing priority species were selected. These priority species were identified based on Freitag & van Jaarsveld's, (1997) Regional Priority Score (RPS) technique. This technique uses a combination of relative rarity, endemicity, vulnerability and taxonomic distinctiveness of each species to determine how important they are within the region (see Freitag & van Jaarsveld., 1997 for formulae and descriptions of the techniques). Grid cells containing the top five and ten percent of these priority species were then identified. Additionally grid cells of biodiversity conservation importance were identified using CPlan (Pressey *et al.*, 1993; Pressey & Logan, 1997; Cowling *et al.*, 1999; Ferrier *et al.*, 2000). These areas are irreplaceable (site irreplaceability = 1) grid cells for a target of 100% species representation i.e. they contain species not recorded anywhere else in the province.

Complementary networks of conservation areas selected by richness-based algorithms (Chapter 2,3,4 and 5), land-use constrained (LUC) algorithms (Chapter 5; Addendum II) and beta diversity (BD) algorithms (Chapter 5) were selected for all bird species. The richness-based algorithm selects a complementary set of grid cells that represents all species at least once. The LUC algorithm does the same, while attempting to avoid areas currently largely transformed and degraded. The BD algorithm also represents all species once in a complementary fashion, but focuses on areas with high turnover in species diversity (beta diversity). These areas were all identified using a 50% level of preselection; this means that species in grid cells which are more than 50% protected by the provincial and national protected area network were assumed to be already represented and were excluded from subsequent selection procedures. The databases used were the most recently updated and revised; these datasets and the outputs may therefore differ from those in previous chapters. Another algorithm, which identifies complementary grid cells required to represent 10% of each vegetation type, was also employed. This vegetation based algorithm also used a 50% level of preselection, taking the percentage of each vegetation type represented within grid cells more than 50% protected into account. These areas were then investigated as to the current and potential land-use threats they face.

*Conservation area prioritisation*

The areas of high biodiversity value identified are all of importance in terms of their biodiversity content. However, due to the limited number of these areas that can be protected immediately some form of priority ranking for conservation action is essential. Using the current and potential land-uses within each of these areas one can investigate which of them need immediate attention. With this in mind, a final set of grid cells, a potential provincial conservation plan, will be identified to complement the existing protected areas using all data available, including species and vegetation type data. This set of grid cells will aim to represent all known biodiversity (all bird, butterfly and mammal species, as well as 10% of each vegetation type) within the region, while at the same time avoiding areas that are largely currently transformed and degraded using a land-use constrained (LUC) algorithm (Chapter 5; Addendum II). This combined algorithm is a species richness-based complementary algorithm. It identifies grid cells containing the most complementary species to ones already selected. These grid cells are then used as preselected grid cells which are then added on to represent 10% of each vegetation type (the species first vegetation based combined algorithm from Chapter 4). During the selection of these grid cells areas of high land use change are avoided (see Chapter 5 & Addendum II).

Each of these grid cells will then be investigated as to their area already transformed, the area suitable for other land-uses and the number of different land-uses that could be practised within that grid cell. The grid cells will be ranked from 1-55 according to increasing levels of suitability for various land-uses, area of road-effects and the number of alternate land-uses for which they are suitable. They will also be ranked from 55-1 according to increasing levels of natural vegetation remaining. Therefore a grid cell with high suitability for a large number of land-uses, a large road-effect zone and little natural vegetation remaining will be ranked close to 55 for all categories. These ranks will be averaged for each grid cell and this will then provide a priority ranking of these areas for conservation action.

## Results

*Land-use impacts*

Table 2 and Figure 3 illustrate that 78.4% of the province is either suitable for alternate forms of land-use or affected by the road-effect zone. Of that area 71.87% is still natural, while 18.4% is already transformed and 9.73% degraded. This already transformed area is made up of cultivated land (86.18%), urban areas (7.75%), forestry (5.48%) and mining (0.5%). Forestry potentially poses one of the largest threats to the area with over 75% of the province being suitable for afforestation, largely through *Eucalyptus camaldulensis* and *E. tereticornis*. Pine and wattle plantations each threaten only about 2.5% of the land. Cultivation, mostly rainfed, is possible in 8.7% of the province.

**Table 2:** Land area potentially suitable for land-uses and/or impacted on by road-effects in the Northern Province.

| General land-use | Land-use type | | Area suitable ($km^2$) | Area suitable (%) | Average area in grid cells (%) |
|---|---|---|---|---|---|
| **Forestry** | | | 93084.35 | 75.65 | 61.04 |
| | *Eucalyptus* | | 93016.64 | 75.59 | 60.99 |
| | | *camaldulensis* | 86793.40 | 70.54 | 56.91 |
| | | *nitens* | 1804.84 | 1.47 | 1.18 |
| | | *saligna* | 4713.29 | 3.83 | 3.09 |
| | | *tereticornis* | 33532.35 | 27.25 | 21.99 |
| | | *urophylla* | 4254.59 | 3.46 | 2.79 |
| | *Pinus* | | 3071.46 | 2.50 | 2.01 |
| | | *elliottii* | 3040.30 | 2.47 | 1.99 |
| | | *patula* | 1151.85 | 0.94 | 0.76 |
| | | *taeda* | 1563.57 | 1.27 | 1.03 |
| | *Acacia* | *mearnsii* | 3055.43 | 2.48 | 2.00 |
| **Cultivation** | | | 10728.95 | 8.72 | 7.04 |
| | Rainfed | | 8144.47 | 6.62 | 5.34 |
| | Irrigated | | 2750.56 | 2.24 | 1.81 |
| **Mining** | | | 1957.42 | 1.59 | 1.29 |
| | Mineral | | 1694.59 | 1.38 | 1.11 |
| | Dimension stone | | 292.43 | 0.24 | 0.19 |
| **Road-effect** | | | 5772.54 | 4.69 | 3.78 |
| **Total area suitable** | | | 96451.94 | 78.38 | 63.95 |

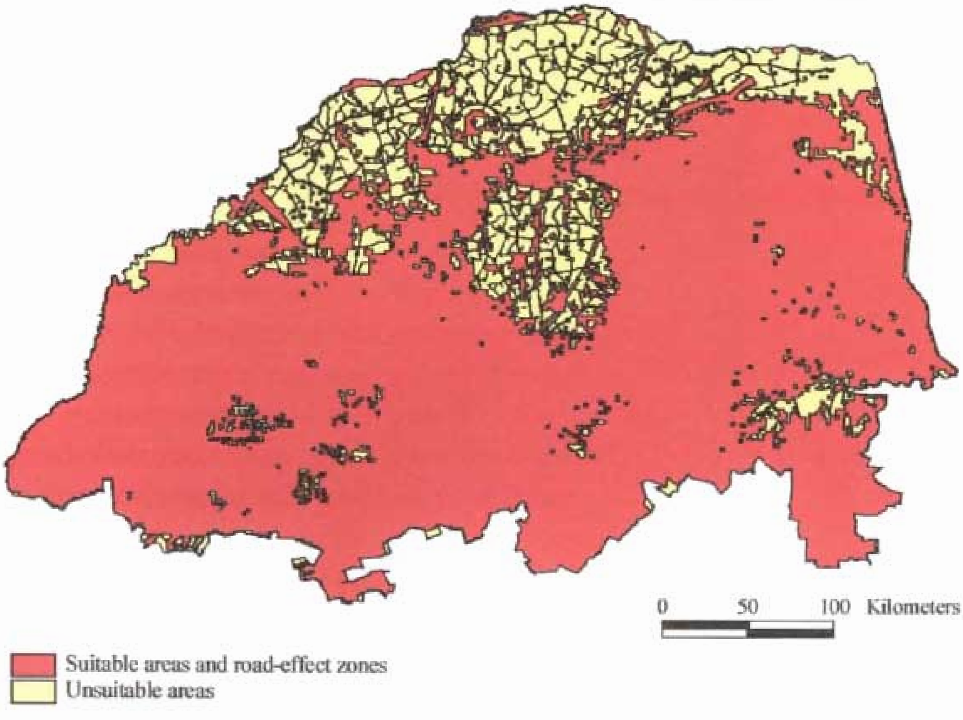


**Figure 3:** Areas suitable for afforestation, cultivation and mining as well as areas within road-effect zones.

Mining and quarrying can potentially occur in 1.6% of the area, while road-effects impact on almost 5% of the province. Table 2 also illustrates the average area within all 15' x 15' grid cells suitable for the various forms of land-use and/or exposed to road-effects. Similar patterns described in the previous paragraph are once again evident. On average 64% of the area within grid cells can be potentially impacted on by land-uses or roads, primarily through commercial forestry, followed by cultivation and finally road-effects and mining. Figure 4 illustrates the current land-cover occurring within areas suitable for alternate land-uses, as well as within the road-effect zones. It is evident that most of the areas suitable for various land-uses are not currently occupied by these specific land-uses. Most of the potential land-uses and road-effects occur in largely natural areas. The second form of existing land-cover type found within these potentially suitable areas is cultivation followed by degraded areas.

Table 3 presents the results of the vegetation analysis. It illustrates the percentage natural area remaining within the vegetation types as well as the percentage of that remaining natural vegetation suitable for alternate land-uses and/or impacted on by road-effects. Most of the vegetation types have large tracts of natural vegetation remaining. Only the Clay Thorn Bushveld, Mixed Lowveld Bushveld, and Sour Lowveld Bushveld contain less than 60% natural vegetation. Most of these largely natural vegetation types are highly suitable for alternate land-uses or are impacted on by road-effects. The Sweet Bushveld, Mopane Bushveld and Soutpansberg Arid Mountain Bushveld are possible exceptions with just over 50% suitable and/or contained within the road-effect zone.

*Biodiversity value*

Table 4 lists all potential priority conservation areas identified in this study, using a wide variety of approaches, along with the number of grid cells they require. Richness hotspots and irreplaceable grid cells require the least amount of land area (4.65%), while rarity hotspots and grid cells containing species with high RPS scores require the most (from 14.9-35.8%). The areas identified are not largely transformed at present, all containing approximately 70% natural vegetation. However, a large proportion of these areas of high biodiversity value is suitable for alternate land-uses. Once again afforestation poses the largest threats, followed by cultivation, road-effects and mining. Similarly, the combined algorithm selects grid cells that are currently largely untransformed but have a high suitability for forestry and cultivation. The area required by this algorithm (25%) is large in comparison with the other areas identified, with the exception of the grid cells representing species with the top 10% RPS scores.

Figure 5 illustrates the grid cells selected by the final combined algorithm based on one representation of all bird, butterfly and mammal species as well as 10% of each vegetation type. This algorithm contained a land-use constraint component and attempted to avoid grid cells largely currently transformed and degraded. The grid cells are colour coded according to their priority rank calculated from the average threat ranks of the grid cells provided in Table 5.

**Table 3:** Remaining natural vegetation within each of the vegetation types of the Northern Province as well as the percentage of that area suitable for alternate land-uses.

| Vegetation type | Remaining natural area (%) | Natural area suitable (%) |
|---|---|---|
| Afromontane Forest | 77.61 | 98.94 |
| Mopane Shrubveld | 100 | 84.51 |
| Mopane Bushveld | 92.05 | 55.79 |
| Soutpansberg Arid Mountain Bushveld | 83.79 | 51.16 |
| Waterberg Moist Mountain Bushveld | 90.24 | 95.21 |
| Lebombo Arid Mountain Bushveld | 98.77 | 87.73 |
| Clay Thorn Bushveld | 48.68 | 99.34 |
| Sweet Bushveld | 78.12 | 51.55 |
| Mixed Bushveld | 65.68 | 85.90 |
| Mixed Lowveld Bushveld | 59.73 | 92.06 |
| Sweet Lowveld Bushveld | 94.37 | 88.26 |
| Sour Lowveld Bushveld | 51.72 | 95.37 |
| Kalahari Plains Thorn Bushveld | 86.03 | 95.69 |
| Moist Sandy Highveld Grassland | 48.28 | 100.00 |
| North-eastern Mountain Grassland | 81.61 | 99.83 |

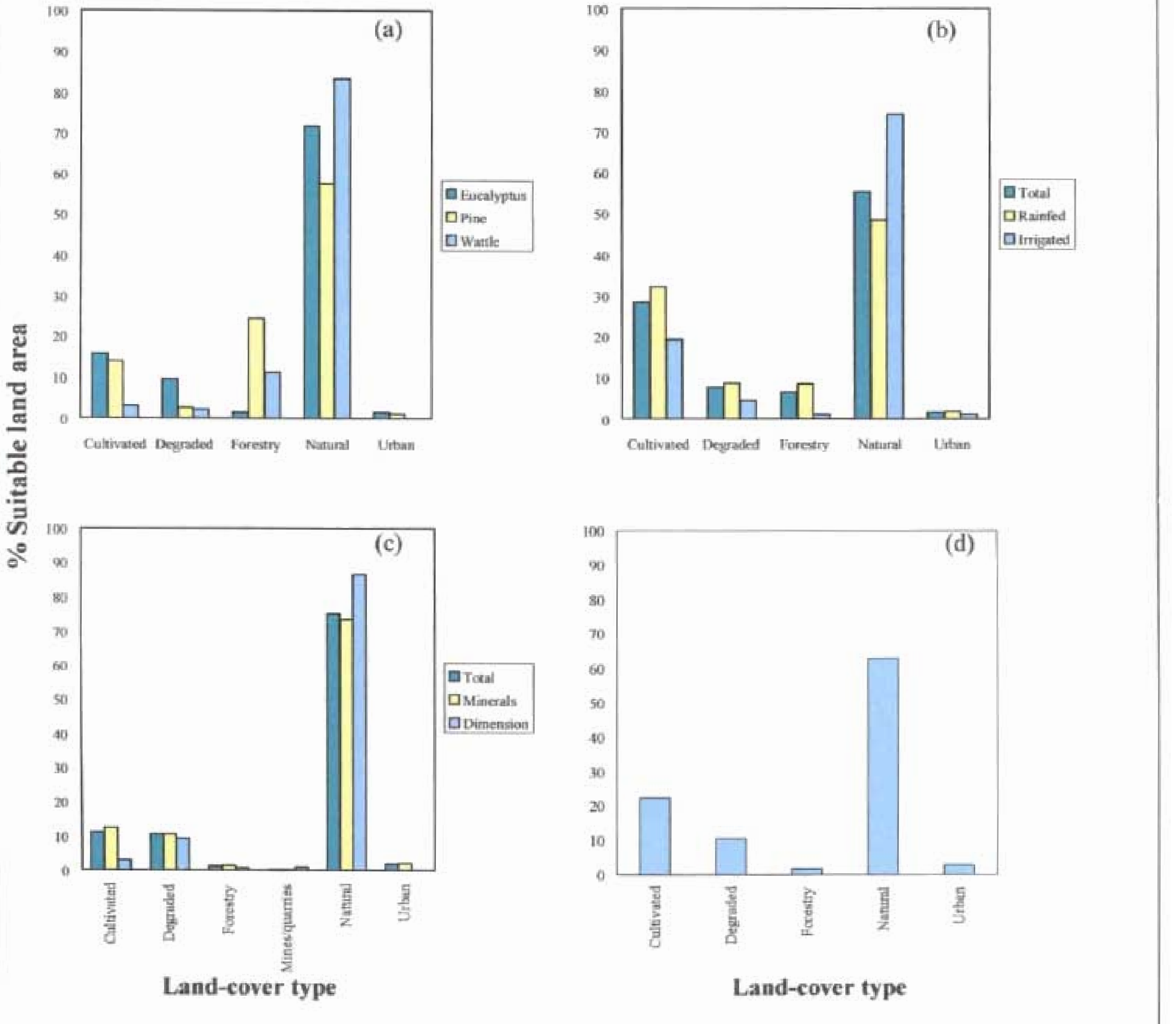


**Figure 4:** An investigation into the current land uses within areas suitable for alternate land uses. An evaluation of the degree of overlap between areas suitable for: (a) forestry, (b) cultivation, (c) mining, or (d) impacted on by road-effect zones, and current land-cover.

**Table 4:** Percentage area required, currently transformed, potentially suitable for a variety of alternative land-uses and impacted on by road-effects within grid cells of high biodiversity value.

| Conservation area | Required | Transformed | Suitable | Forest | Eucalyptus | Pine | Wattle | Cultivated | Rainfed | Irrigated | Mine | Mineral | Dimension stone | Road-effect |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Richness hotspot | 4.65 | 38.63 | 62.13 | 60.11 | 60.11 | 12.73 | 7.40 | 19.83 | 19.45 | 0.47 | 0.62 | 0.57 | 0.05 | 7.63 |
| Rarity hotspot | 14.88 | 31.67 | 64.18 | 62.06 | 62.05 | 7.47 | 5.39 | 12.68 | 10.64 | 2.12 | 1.38 | 1.34 | 0.04 | 4.70 |
| RPS (5%) | 17.67 | 27.93 | 69.36 | 67.97 | 67.76 | 3.96 | 3.06 | 8.08 | 7.35 | 0.77 | 1.05 | 0.96 | 0.12 | 4.56 |
| RPS (10%) | 35.81 | 31.30 | 69.43 | 67.49 | 67.38 | 4.59 | 4.22 | 9.82 | 8.40 | 1.66 | 1.30 | 1.24 | 0.09 | 4.51 |
| Irreplaceable sites | 4.65 | 29.05 | 42.62 | 41.55 | 41.55 | 1.07 | 0.62 | 3.01 | 1.35 | 1.66 | 0.79 | 0.75 | 0.03 | 2.74 |
| Richness algorithm | 7.91 | 26.69 | 46.84 | 45.48 | 45.48 | 5.56 | 1.73 | 9.71 | 8.73 | 1.03 | 0.93 | 0.89 | 0.04 | 4.51 |
| LUC algorithm | 9.30 | 23.89 | 54.17 | 52.40 | 52.37 | 5.67 | 3.74 | 7.10 | 5.50 | 1.64 | 1.21 | 1.17 | 0.04 | 3.90 |
| Beta diversity algorithm | 11.16 | 25.47 | 61.50 | 58.81 | 58.81 | 4.34 | 1.44 | 6.51 | 4.67 | 1.87 | 1.23 | 1.18 | 0.06 | 4.68 |
| Vegetation algorithm | 9.30 | 22.85 | 60.77 | 57.88 | 57.88 | 1.88 | 2.64 | 8.67 | 6.71 | 2.38 | 1.60 | 1.03 | 0.63 | 4.23 |
| **Combined algorithm** | **25.58** | **26.05** | **66.00** | **63.61** | **63.52** | **5.34** | **4.87** | **10.00** | **8.30** | **1.85** | **1.45** | **1.33** | **0.15** | **4.78** |

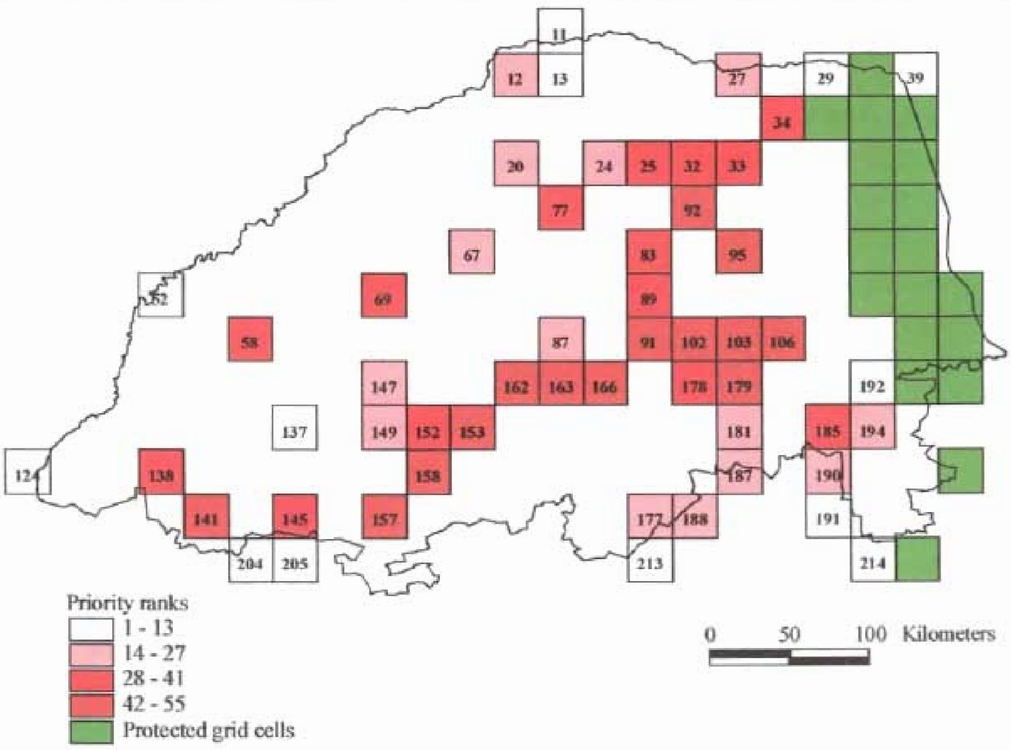


**Figure 5:** Priority ranks due to land-use threats facing grid cells selected by a species and vegetation combined algorithm including a land-use constraint. Numbers in grid cells represent grid cell codes (see Column 1 in Table 5). High ranks (i.e. dark red) illustrate high priority.

Table 5: Grid cells selected by combined algorithm and the ranked threats they face. The highest priority rank (55) denotes the grid cell under highest threat.

| Grid cell code | Natural | Suitability | Cultivated | Mining | Forestry | Road-effect | Number of threats | Average threat rank | Priority rank |
|---|---|---|---|---|---|---|---|---|---|
| 179 | 45 | 52 | 49 | 54 | 50 | 43 | 45 | 48.3 | 55 |
| 153 | 39 | 50 | 47 | 50 | 44 | 53 | 54 | 48.1 | 54 |
| 157 | 46 | 41 | 48 | 47 | 40 | 55 | 55 | 47.4 | 53 |
| 102 | 52 | 54 | 55 | 13 | 52 | 46 | 48 | 45.7 | 52 |
| 92 | 40 | 51 | 52 | 36 | 55 | 39 | 41 | 44.9 | 51 |
| 163 | 29 | 49 | 21 | 53 | 48 | 52 | 53 | 43.6 | 50 |
| 162 | 23 | 48 | 40 | 44 | 47 | 49 | 50 | 43.0 | 49 |
| 91 | 55 | 55 | 53 | 12 | 51 | 47 | 21 | 42.0 | 48 |
| 103 | 48 | 34 | 54 | 32 | 34 | 45 | 47 | 42.0 | 47 |
| 95 | 51 | 53 | 34 | 43 | 54 | 21 | 29 | 40.7 | 46 |
| 158 | 30 | 42 | 25 | 46 | 41 | 48 | 49 | 40.1 | 45 |
| 106 | 18 | 44 | 51 | 31 | 46 | 44 | 46 | 40.0 | 44 |
| 141 | 28 | 40 | 45 | 49 | 35 | 38 | 40 | 39.3 | 43 |
| 166 | 41 | 43 | 41 | 38 | 43 | 24 | 31 | 37.3 | 42 |
| 33 | 43 | 37 | 50 | 29 | 42 | 27 | 32 | 37.1 | 41 |
| 152 | 37 | 39 | 39 | 30 | 39 | 37 | 39 | 37.1 | 40 |
| 83 | 31 | 27 | 22 | 51 | 24 | 50 | 51 | 36.6 | 39 |
| 32 | 36 | 31 | 43 | 48 | 31 | 28 | 33 | 35.7 | 38 |
| 185 | 33 | 28 | 42 | 22 | 28 | 42 | 44 | 34.1 | 37 |
| 145 | 24 | 30 | 32 | 55 | 30 | 32 | 35 | 34.0 | 36 |
| 89 | 35 | 21 | 44 | 15 | 22 | 41 | 43 | 31.6 | 35 |
| 25 | 17 | 22 | 29 | 24 | 23 | 51 | 52 | 31.1 | 34 |
| 58 | 2 | 45 | 19 | 35 | 53 | 29 | 34 | 31.0 | 33 |
| 178 | 9 | 46 | 38 | 42 | 49 | 8 | 24 | 30.9 | 32 |
| 34 | 44 | 23 | 37 | 45 | 21 | 17 | 28 | 30.7 | 31 |
| 77 | 38 | 20 | 31 | 20 | 19 | 40 | 42 | 30.0 | 30 |
| 69 | 22 | 25 | 16 | 41 | 27 | 36 | 38 | 29.3 | 29 |
| 138 | 13 | 38 | 46 | 17 | 38 | 22 | 30 | 29.1 | 28 |
| 181 | 16 | 47 | 33 | 21 | 45 | 10 | 26 | 28.3 | 27 |
| 87 | 42 | 26 | 9 | 18 | 25 | 54 | 22 | 28.0 | 26 |
| 177 | 54 | 29 | 10 | 25 | 29 | 23 | 17 | 26.7 | 25 |
| 67 | 53 | 32 | 11 | 23 | 32 | 19 | 15 | 26.4 | 24 |
| 147 | 19 | 35 | 13 | 33 | 37 | 26 | 18 | 25.9 | 23 |
| 24 | 5 | 19 | 35 | 27 | 17 | 35 | 37 | 25.0 | 22 |
| 194 | 10 | 36 | 28 | 10 | 36 | 34 | 20 | 24.9 | 21 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 187 | 27 | 17 | 36 | 16 | 18 | 9 | 25 | 21.1 | 20 |
| 149 | 25 | 33 | 12 | 6 | 33 | 30 | 8 | 21.0 | 19 |
| 20 | 12 | 18 | 7 | 39 | 20 | 31 | 19 | 20.9 | 18 |
| 12 | 1 | 11 | 24 | 28 | 10 | 33 | 36 | 20.4 | 17 |
| 188 | 21 | 13 | 26 | 26 | 13 | 6 | 23 | 18.3 | 16 |
| 27 | 7 | 9 | 20 | 52 | 5 | 16 | 14 | 17.6 | 15 |
| 190 | 32 | 14 | 23 | 9 | 14 | 15 | 13 | 17.1 | 14 |
| 52 | 14 | 16 | 15 | 19 | 16 | 12 | 27 | 17.0 | 13 |
| 29 | 20 | 7 | 27 | 37 | 3 | 13 | 11 | 16.9 | 12 |
| 192 | 8 | 15 | 6 | 34 | 15 | 20 | 16 | 16.3 | 11 |
| 191 | 50 | 4 | 17 | 8 | 8 | 4 | 9 | 14.3 | 10 |
| 11 | 11 | 8 | 30 | 11 | 7 | 14 | 12 | 13.3 | 9 |
| 204 | 34 | 3 | 18 | 14 | 2 | 11 | 10 | 13.1 | 8 |
| 137 | 4 | 24 | 8 | 5 | 26 | 18 | 6 | 13.0 | 7 |
| 13 | 3 | 6 | 3 | 40 | 4 | 25 | 7 | 12.6 | 6 |
| 214 | 49 | 1 | 14 | 7 | 1 | 1 | 1 | 10.6 | 5 |
| 205 | 47 | 5 | 2 | 2 | 9 | 5 | 4 | 10.6 | 4 |
| 213 | 26 | 10 | 4 | 3 | 11 | 3 | 3 | 8.6 | 3 |
| 124 | 15 | 12 | 5 | 4 | 12 | 7 | 5 | 8.6 | 2 |
| 39 | 6 | 2 | 1 | 1 | 6 | 2 | 2 | 2.9 | 1 |

Grid cells with high priority ranks have a combination of high suitabilities for a large number of alternate land-uses, large road-effect zones and low levels of current natural land-cover extent. Appendix 1 provides the raw values used for these rank calculations in Table 5. Grid cells of high priority are concentrated in the central and southern regions (Figure 5). These areas are currently largely cultivated, urbanised and degraded (Figure 1) and are also suitable to most forms of land-use (Figure 2).

**Discussion**

*Future land-use scenarios*

The Northern Province is a largely untransformed area with over 70% of the region still covered by natural vegetation. When this is compared with other regions, where only 34% (Hokitika, New Zealand; Awimbo *et al.,* 1996), 8% (Bega Valley, New South Wales; Keith, 1995) or as little as 7% natural vegetation remains (Western Australian wheatbelt; Saunders *et al.*, 1993), the biodiversity in the province does not appear to be in the dire situation prevailing elsewhere. However, this low level of transformation within the province is no reason for complacence. First, this estimate gives no indication of how intact or fragmented that remaining vegetation is; some of the natural pieces of land left may have lost their ability to sustain biodiversity and ecological processes. This would depend on the susceptibility of individual species to extirpation, local scale landscape patterns, the nature and environmental impact of interspersed alternative land-uses, the impact of e.g. agricultural or forestry practices on hydrological processes and soil properties, and degradation within natural areas (Saunders *et al.*, 1993; Scholtz & Chown, 1993; Freemark, 1995; Allan *et al.*, 1997; White *et al.,* 1997; Brokaw, 1998; van Jaarsveld *et al.*, 1998a; Joubert, 1998; Seymour, 1998).

Second, the fact that the land is as yet undeveloped is not due to its unsuitability for alternate land-uses. As illustrated by the results there are substantial tracts of land suitable for a variety of largely destructive land-uses. Almost 80% of the province can be used for some form of afforestation, cultivation and mining or quarrying. With increasing population sizes and their associated resource and land-use demands those areas currently under some form of cultivation, forestry or other land-use are likely to increase. South Africa has already witnessed an increase of 50.5 and 7.5% in areas under afforestation and cultivation, respectively, since the mid-1980's up to the 1994 National Land-cover database estimates (Fairbanks *et al.*, 2000).

Forestry

The Northern Province is part of the major commercial forestry area in South Africa and the current levels of afforestation (0.81%) could increase substantially with the use of specialised species (Thompson, 1995; Fairbanks, 1997). With almost 75% of the province being suitable for forestry, as well as the fact that the forestry sector has been identified as one of the sectors that can provide additional employment and financial resources in South Africa, the potential impact of forestry on regional biodiversity is cause for concern. The South African Reconstruction and Development Program (RDP) strategy identifies the forest sector as an important element of local natural resources development that can contribute to creating better living environments and economic opportunity. The forest and forest products industry is a major employer and of great importance to the South African labour market. It is estimated that about 200000 to 260000 people are employed in the forest and wood processing industries. An estimated 120000 people are employed in those industries which use wood as a primary input (DWAF, 1996). In addition, there are those employed by the smaller primary converters such as in making poles, matches and charcoal.

Industrial forestry began in the last quarter of the nineteenth century and has proved to be highly profitable in the use of natural resources, although it comes at an environmental and social cost (DWAF, 1996). By 1994, industrial forests in South Africa had grown to about 1,45 million hectares. Of the planted areas, 56% was pines, 32% was eucalyptus and 11% was wattle. New afforestation has increased the total area of plantation by about 17000 ha per year recently (DWAF, 1996). This has been supported by past Government policy to expand the plantations, and by the need for wood for the pulp and paper sector. New afforestation has slowed down, however, with very few permits having been issued in the last year (DWAF, 1996).

Potential productivity of these forests is relatively high by world standards, which satisfies over 90% of domestic demand and provides for a surplus for export, largely as pulp, paper, wood chips and other products. The forest products industries, i.e. all those industries using wood and wood products as raw material, constitute a significant part of the South African economy, contributing about 7,4% to the output of the country's manufacturing sector in 1993/94 (DWAF, 1996). They earned about R1,28 billion in net foreign exchange from total export earnings of about R3,6 billion in 1994/95. Their relative contribution to the economy has grown steadily in the past 20 years. The many jobs involved in these industries mean that over one million mainly rural people depend on this industry directly (DWAF, 1996).

Cultivation

Cultivated land has steadily increased in South Africa since the turn of the century from approximately three percent in 1911 to eight percent in 1981 and finally to 12.11% in 1994 (Scotney *et al.*, 1988; Fairbanks *et al.*, 2000). Agriculture is an important primary component in the South African economy as well as for the community. Not only is agriculture often the major factor in rural economic growth and development, but the necessary programmes to support agriculture play a distinctive role in broadening the economic and social options of rural and urban people, and consequently in improving their quality of life (NDA, 1995). The contribution of primary agriculture to the South African GDP declined from a level of 12% in 1960 to approximately 5% in 1990. Since 1994 this contribution has varied between 3 and 4% of the GDP. This relative decline in the nominal contribution to GDP does not imply that the agricultural GDP has declined in real terms, but that there was a faster growth in other sectors. In 1998 the contribution of primary agriculture to the GDP amounted to R21 607 million, which represented 3,2% of GDP (NDA, 2000). It should be added that with strong linkages to the rest of the economy, the "agro-industrial" complex is estimated to contribute at least 15% of the GDP (NDA, 2000).

In their review of agricultural production for 1999/2000 the National Directorate of Agriculture (NDA, 2000) state that South Africa has for some time been one of the few countries in the world that are net exporters of agricultural produce. In the period 1994 to 1998 the agricultural contribution to total export values was in the order of 8 to 10%. The agricultural share in total imports varied between 6 and 7% during the same period. Exports exceeded the value of imports during this period by percentages which varied between 19 (1995) and more than 60 (1998). Agriculture has also long been the sector with the largest formal wage employment in the South African economy. In 1970 30,6% of the economically active population was employed in the agricultural sector. Even though this percentage declined to 13,2% in 1994, it still represented 1,28 million jobs. With only 3,3 million workers in the rural areas of South Africa, this means that agriculture provided for almost 40% of formal employment in rural areas.

<u>Forestry and cultivation scenarios</u>

The Northern Province has large areas of mostly untransformed arable land as illustrated in this study. However, as evidenced by the results there is a lot of overlap between areas suitable for forestry and cultivation. Suitable agricultural land is limited due to scarce water resources and suitable land of high agricultural potential. There is also an increased demand for land by non-agricultural land-uses including residential and industrial development. In addition to this economic development and national food security depend on the availability of productive and fertile agricultural land. The White Paper on Agriculture (NDA, 1995) therefore states that it is imperative for agriculture to utilise these two resources to ensure the sustainable production of agricultural products. Thus South Africa's productive agricultural land should be retained for agricultural use and the use of agricultural land for other purposes should be minimised. This trade-off and land-use decision making with respects to these two large land-uses of forestry and agriculture will depend on the policies and economics of the day.

Limiting factors for the further expansion of forestry and cultivation within the province include the low rainfall and new water laws (DWAF, 1996). The aridity of the province has up till now limited the afforested areas to the moist escarpment and mountain regions (see Figure 4 in Chapter 1) and the commercial cultivated areas to the same areas as well as the Springbok flats. However the development of specialised species with higher levels of drought and frost resistance has increased the afforestable areas within the province (Figure 2) (Fairbanks, 1997). In opposition to this expansion, the South African National Water Act (Act 36 of 1998) now includes a section on stream flow reduction activities. This allows for the regulation of land-based activities that reduce stream flow. These activities include the use of land for afforestation which has been or is being established for commercial purposes. Other activities including the cultivation of any particular crop or other vegetation can also be declared to be a stream flow reduction activity. Therefore although the areas under forestry and cultivation will in all likelihood be more intensively farmed and expanded in South Africa in order to meet the needs of expanding future generations, the aridity of the province as well as the water laws will in probability constrain this expansion. Stream flow reductions and failure of irrigation schemes all contribute to the fact that these two forms of land-use will not expand and may in fact recede.

Alternate land-uses

The province's wide variety of mineral and dimension stone wealth is clear, as well as the potential implications this would have for regional biodiversity. However this wealth and impact may be underestimated within the current study due to the limited number of areas that have been explored within the province. Many mineral regions and provinces within the study area remain unexplored (Wilson & Anhaeusser, 1998). Finally the impacts of roads on the biodiversity appears to be consistent with levels of impacts found in the rest of the country at approximately 5% of the region being impacted (Addendum I). Although these road-effect zones are useful ways to approximate the threats facing regions due to road infrastructure they may not be the best indicators and more work is required on determining the impacts of roads on biodiversity. As Stoms (2000) points out, many aspects of roads affect biodiversity: road width, traffic volume, traffic speed, vehicle miles travelled, road network structure or its spatial configuration, management of the right-of-way, noise levels, light disturbance, and chemical pollution. Most of these factors vary over daily, weekly and annual cycles, which may interfere with critical behavioural periods such as breeding or migration. As such, the road-effect zone can represent only a first order approximation attempt to capture more of the multi-dimensional nature of road network effects

*Impacts of future land-uses*

Although all of these land-uses pose a threat to regional biodiversity through land-use transformation and alteration and were treated as equally important threats, it is important to note that the types and magnitude of the impacts they have on biodiversity will differ widely. Both afforestation and cultivation, the main threats to biodiversity within the province, are monocultures usually planted on a large scale. However their impacts on biodiversity structure and function are very different. Afforestation involves the replacement of natural vegetation such as grassland or woodland, as well as ancient communities rich in species (DWAF, 1996). Fundamental habitat changes of this kind obviously impact upon biodiversity (Allan *et al.*, 1997). This is especially the case for transformation of areas that were open grassland or woodland areas (like those in the Northern Province) to closed-canopy plantations of alien trees resulting in large-scale habitat changes. Afforestation also has biological implications and has important impacts on regional biodiversity including decreases and changes in species community diversity (Armstrong &

van Hensbergen, 1994; 1995; 1996; 1997; Armstrong *et al.*, 1998) especially of globally and regionally threatened species (Allan *et al.*, 1997). The disruption of natural ecological processes of normally open grasslands and woodlands e.g. fire regimes, species movements, hydrological and nutrient cycles, is also enormous (Richardson & van Wilgen, 1986; Saunders *et al.*, 1991).

Most industrial forests in South Africa were established in grassland ecosystems on naturally acid soils which are prone to loss of mineral nutrients. Where mineral nutrients in the wood are exported by harvesting, or if the forest litter is not effectively recycled, the already acid soils lose fertility. The combination of acidification and forestry effects has been found to be comparable to areas affected by 'acid rain' to the worst degree in industrialised countries (DWAF, 1996). The loss of nutrients is worsened by the increasing acidity of rainfall over much of the region, caused principally by industrial pollution. Afforestation also has serious water budget economic and sociological implications in South Africa as well as in other regions of the world (Macdonald, 1989; National Water Act, (Act 36 of 1998)). An example of these broader-scale effects is the impact on water in rivers which flow through protected areas, such as the Kruger National Park. This, together with other factors such as irrigation abstraction and prolonged droughts, have jeopardised aquatic ecosystems (DWAF, 1996). The forestry and forest products industries also have other environmental impacts which must be recognised and managed. Sawmills, mining timber mills, pulp and paper mills generate waste and water- and airborne emissions which are environmentally harmful and often offensive to neighbouring people.

The impacts of agriculture are argued to be less far-reaching and threatening to biodiversity. Structural changes are relatively minor in comparison, water demands are less, and landscapes fragmented by areas under cultivation still allow for dispersal of plant and animal species (Freemark, 1995). There are however some impacts that must be considered and managed within cultivated areas. Cultivated areas do disrupt natural ecosystems and their processes (DEAT, 1996). Agro-pesticides and herbicides have been shown to have negative impacts on various animal species (Freemark, 1995). The intensification and abandonment of traditional farming methods have also resulted in declines in biodiversity (Suárez *et al.*, 1997). Hinsley *et al.* (1998) point out that abandonment of fields can result in plant species invasion and subsequent biodiversity declines.

Mining, although its impacts are felt on a finer spatial scale, can result in drastic habitat changes through mine dumps, pollution of ground and surface water, ground and air pollution, alien plant invasions, erosion and topsoil and vegetation losses. The ecological effects of roads mentioned previously are indications of the far reaching impacts of roads on biodiversity. Therefore it is important that although all of these land-uses were considered to be equally important in threatening biodiversity, any conservation or land-use planning initiative within the province would have to consider the differing impacts of these land-uses.

Another form of land-cover threat in the province, one that is not easily mapped or predicted in studies such as this one, is land degradation (Scholtz & Chown, 1993; Hoffman, 1999). Land degradation refers to the loss of primary production of natural vegetation in an area. This includes all regions with very low vegetation cover in comparison with the surrounding natural vegetation cover and are typically associated with subsistence level farming and rural population centres, where wood-resource removal, overgrazing and subsequent soil erosion are excessive (Thompson, 1996). South Africa's natural resources are being degraded at an alarming rate (Newby & Wessels, 1997; Hoffman *et al.*, 1999). Soil erosion, as a consequence of overgrazing and improper cultivation, is considered to be one of the most serious environmental problems facing South Africa (Newby & Wessels, 1997; Hoffmann *et al.*, 1999). It is postulated that between 50 and 87% of natural rangeland is in a poor to critical condition (Newby & Wessels, 1997), while an estimated 150 million tons of sediment are transported annually by South African rivers (DEA, 1992).

The Northern Province is made particularly vulnerable to degradation by two factors. First, its high levels of aridity make it vulnerable to the loss of natural vegetation and subsequently to soil degradation which in essence makes the degradation process irreversible and permanent (desertification) (Newby & Wessels, 1997). Second, a large proportion of the human population in the Province is rural and leads a subsistence level lifestyle. Through fuel wood harvesting and domestic livestock grazing on areas particularly susceptible to degradation, thereby exacerbating the process. Other land-uses within the area include commercial domestic and wildlife livestock ranching (land-uses considered to be more amenable to biodiversity conservation) (Pressey, 1992). However, if not managed these land-uses can also degrade this sensitive land (Newby & Wessels, 1997). Therefore, although various acts, laws and permit systems regulate some of the land-use impacts in an area, something must be done to address the ongoing land degradation. Because as Lewis and Berry (1988), and James (1991) predict: if the current rate of degradation continues and sustainable utilisation is not achieved, all attempts at socially and economically uplifting South Africa's people are doomed to fail.

*Land-use planning*

Generally, the use of land in South Africa has been poorly planned, with resultant inefficiencies, inequities, and environmental degradation. Although the most glaring consequences arise from the apartheid policies as applied in the former homelands, effects are evident throughout the rest of the country (DWAF, 1996). Some consequences of inadequate land-use planning are seen in land disputes, the conflicts over water resources, a concern over the loss of land suited to crop cultivation and the loss of habitats for native species. It is obvious from the discussion above that there are several factors that must be taken into consideration during land-use planning. The economic and social implications of various land-uses, are just two of the considerations that must be taken in this decision-making process. The impacts of these land-uses on biodiversity are substantial and often irreversible. As signatories to the Convention on Biological Diversity South Africa is compelled to review the impact of land-use changes on biodiversity and seek changes where necessary. This study therefore provides an important first step towards achieving this goal in that it highlights areas of extreme importance to biodiversity conservation.

At a broad-scale the results illustrate that similar to the findings in Addendum I most of the vegetation types of South Africa are not under large threats at present. However, the Clay Thorn Bushveld and Mixed Lowveld Bushveld are threatened due to high levels of transformation within South Africa as well as within the province and require conservation attention. The fact that these vegetation types and many of the others face large future threats due to high levels of suitability for various land-uses is important and highlights their conservation needs and should be considered in any land-use plan for the area. The Clay Thorn Bushveld as well as the forest and grassland vegetation types are almost entirely suitable for some form of land-use development. This has very important implications for biodiversity conservation.

The grassland biome is an endangered biome in South Africa requiring urgent conservation attention. It is the most productive agricultural area, hosts most of the human population of South Africa, is very rich in minerals (especially coal and gold) and also includes most of the areas suitable to forestry. It is also one of the most important biomes in terms of its biodiversity content due to high levels of species endemism, rarity and richness (Allan *et al.*, 1997; van Jaarsveld *et al.*, 1998a)

Indigenous subtropical forest is the smallest biome represented in southern Africa, covering less than 0.25% of the total land area (Low & Rebelo, 1996), yet it supports a high proportion of the region's floral and faunal diversity (e.g., 14% of all terrestrial birds and mammals; Geldenhuys & MacDevette 1989). The importance of conserving forest biodiversity in southern Africa is widely recognised (e.g., Cooper, 1985; Geldenhuys & MacDevette, 1989; Lawes *et al.*, In Press). However, it is evident that the forest biome is under increasing and harmful anthropogenic pressure (Cunningham, 1989; Geldenhuys & Macdevette, 1989). Macdonald (1989) estimates that approximately 42.5% of the forest biome in South Africa has already been transformed, mostly in the recent past. These results therefore emphasise that the identification of conservation areas in the forest and grassland biome within the region is an issue of immediate concern.

Similarly most of the regions identified as important to biodiversity conservation (hotspots, complementary networks etc.) due to their biodiversity composition are currently not largely threatened by alternate land-uses, but may well become so in the near future due to high levels of suitability to especially forestry. In both the White Papers for forestry and agriculture (NDA, 1995; DWAF, 1996) it is widely acknowledged that demands for both forestry and agricultural products will increase in the future as populations increase, therefore these suitable areas must be seen as real potential threats to areas important to biodiversity.

Finally this study makes a contribution to land-use planning especially conservation planning. These land-use threats to biodiversity, both current and future, are real and serious (Soulé, 1991; Sala *et al.*, 2000). One method of addressing these threats is through the conservation of biodiversity within areas that are protected from these land-use impacts (Margules & Pressey, 2000). However, with the limited resources on hand for conservation area establishment it is obvious that first, not all areas identified as being important to biodiversity conservation will be protected immediately, and second many of these areas may well have to rely on off-reserve management rather than formal protection (Pressey & Logan, 1997, Cowling *et al.*, 1999). The techniques highlighted within this study provide a useful way of deciding which of these areas should receive some form of protection first. The combined conservation area selection algorithm identified grid cells that are arguably all important to effective biodiversity conservation within the province, but by looking at the threats facing those areas one can form an idea of the conservation urgency of some of them. Thus the use of not only the current levels of threat facing biodiversity, but also the incorporation of future threats within the region, allows for effective and efficient conservation planning. One can decide which of the areas if they were to remain unprotected would either lose all the biodiversity or become transformed due to a high land-use suitability and would subsequently lose their biodiversity at some later stage.

**Conclusion**

Sustainable development calls for development that meets the needs of current generations without compromising the needs of future ones (WCED, 1987). This implies improving the quality of life for humans while living within the carrying capacity of the environment (IUCN, 1991). Often though the reality of the situation is that the primary focus is on development with conservation concerns acting as a constraint. However, effective sustainability will only be attainable when a compromise between these two seemingly mutually exclusive forms of land-use can be attained. Regional sustainability will depend on both effective land allocations for alternative and often-competing land-uses and also management of these areas to satisfy multiple goals (Faith & Walker, 1996). We feel this study brings us closer to the goal of regional sustainability in that it acknowledges the need for expansion of current land-use practices, as well as investigating their potential spatial extent and implications for regional biodiversity (at a broad-scale). Finally it admits that the delay between conservation planning and implementation is often lengthy. By ranking the areas of crucial biodiversity importance in order of the number and degree of threat they do and will face, biodiversity losses can be minimised.

**References**

Allan, D.G., Harrison, J.A., Navarro, R.A., van Wilgen, B.W. & Thompson, M.W. (1997). The impact of commercial afforestation on bird populations in Mpumalanga Province, South Africa - insights from bird-atlas data. *Biological Conservation* 79, 173-185.

Allsopp, N. (1999). Consequences to soil patterns and processes in southern Africa following vegetation change associated with livestock management. In *Proceedings of the VI$^{th}$ International Rangelands Congress.* (Eds. D. Eldgridge & D. Freudenberger), Vol. 1, pp 96-100. Townsville, Australia.

Armstrong, A.J. & van Hensbergen, H.J. (1994). Comparison of avifaunas in *Pinus radiata* habitats and riparian habitat at Jonkershoek. *South African Journal of Wildlife Research* 24, 48-55.

Armstrong, A.J. & van Hensbergen, H.J. (1995). Effects of afforestation and clear-felling on birds and small mammals at Grootvadersbosch, South Africa. *South African Forestry Journal* 174, 17-21.

Armstrong, A.J. & van Hensbergen, H.J. (1996). Impacts of afforestation with pines on assemblages of native biota in South Africa. *South African Forestry Journal* 175, 35-42.

Armstrong, A.J. & van Hensbergen, H.J. (1997). Evaluation of afforestable montane grasslands for wildlife conservation in the north-eastern Cape, South Africa. *Biological Conservation* 81, 179-190.

Armstrong, A.J., Benn, G., Bowland, A.F., Goodman, P.S., Johnson, D.N., Maddock, A.H. & Scott-Shaw, C.R. (1998). Plantation forestry in South Africa and its impact on biodiversity. *South African Forestry Journal* 182, 59-65.

Awimbo, J.A., Norton, D.A. & Overmars, F.B. (1996). An evaluation of representativeness for nature conservation, Hokitika ecological district, New Zealand. *Biological Conservation* 75, 177-186.

Balmford, A., Mace, G.M. & Ginsberg, J.R. (1998). The challenges to conservation in a changing world: putting processes on the map. In *Conservation in a changing world* (Eds. G.M. Mace, A. Balmford & J.R.Ginsberg), pp 1-28. Cambridge University Press, UK.

Brokaw, N. (1998). Fragments past present and future. *Trends in Ecology & Evolution* 13, 382-383.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. & Diaz, S. (2000). Review article: Consequences of changing biodiversity. *Nature* 405, 234-242.

Cooper KH (1985). *The Conservation Status of Indigenous Forests in Transvaal, Natal and O.F.S., South Africa*. Wildlife Society of South Africa, Durban.

Cowling, R.M., Pressey, R.L., Lombard, A.T., Desmet, P.G. & Ellis, A.G. (1999). From representation to persistence: requirements for a suistainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa. *Diversity & Distributions* 5, 51-71.

Cunningham, A.B. (1989). Indigenous plant use: balancing human needs and resources. In *Biotic Diversity in Southern Africa* (Ed B.J. Huntley), pp 93-106. Oxford University Press, Cape Town.

DEA. (1992). *Building the foundation for sustainable development in South Africa: National report to United Nations Conference on Environment and Development (UNCED), Rio de Janeiro.* Department of Environmental Affairs, Pretoria, South Africa.

DEAT. (1996). *National Protected Areas Database.* Department Environmental Affairs and Tourism, Pretoria.

DWAF. (1996). *White paper on forestry: sustainable forest development in South Africa: the policy of the government of national unity.* Department of Water Affairs and Forestry, Pretoria.

ESRI. (1998). *Arc and Grid Command References.* Environmental Systems Research Institute, Redlands, California.

Fairbanks, D.H.K. & Thompson, M.W. (1996). Assessing land-cover map accuracy for the South African land-cover database. *South African Journal of Science* 92, 465-470.

Fairbanks, D.H.K. (1997). Regional *Eucalyptus* species suitability mapping: Western Cape and hybrids for summer rainfall zones. Environmentek, CSIR, Pretoria. DWAF Report ENV/P/C 97271.

Fairbanks, D.H.K., Thompson, M.W.M., Vink, D.E., Newby, T. S., van den Berg, H.M. & Everard, D.A. (2000). The South African land-cover characteristics database: a synopsis of the landscape. *South African Journal of Science* 96, 69-82.

Faith, D.P. (1995). *Biodiversity and regional sustainability analysis.* CSIRO, Division of Wildlife and Ecology, Canberra, Australia.

Faith, D.P. & Walker, P.A. (1996). Environmental diversity: on the best possible use of surrogate data for assessing the relative biodiversity of sets of areas. *Biodiversity & Conservation* 5, 399-415.

Ferrier, S., Pressey, R.L. & Barrett, T.W. (2000). A new predictor of the irreplaceability of areas for acheiving a conservation goal, its application in real-world planning, and a research agenda for further refinement. *Biological Conservation* 93, 303-325.

Forman, R.T.T. (2000). Estimate of the area affected ecologically by the road system in the United States. *Conservation Biology* 14, 31-35.

Forman, R.T.T. & Alexandra, L.E. (1998). Roads and their major ecological effects. *Annual Reviews of Ecology & Systematics* 29, 207-231.

Freemark, K. (1995). Assessing effects of agriculture on terrestrial wildlife: developing a hierarchical approach for the US EPA. *Landscape & Urban Planning* 31, 99-115.

Freitag, S. & Mansell, M.W. (1997). The distribution and protection status of selected antlion species (Neuroptera: Mymeleontidae) in South Africa. *African Entomology* 5, 205-16.

Freitag, S. & van Jaarsveld, A.S. (1995). Towards conserving regional mammalian species diversity: a case study and data critique. *South African Journal of Zoology* 30, 136-144.

Freitag, S. & van Jaarsveld, A.S. (1997). Relative occupancy, endemism, taxonomic distinctiveness and vulnerability: prioritizing regional conservation actions. *Biodiversity & Conservation* 6, 211-232.

Freitag, S., Nicholls, A.O. & van Jaarsveld, A.S. (1998). Dealing with established reserve networks and incomplete distribution data sets in conservation planning. *South African Journal of Science* 94, 79-86.

Geldenhuys, C.J. & MacDevette, D.R. (1989). Conservation status of coastal and montane evergreen forest. In *Biotic Diversity in Southern Africa* (Ed. B.J. Huntley), pp 224-238. Oxford University Press, Cape Town.

Harrison, J.A. (1992). The South African Bird Atlas Project: Five years of growth. *South African Journal of Science* 88, 410-413.

Hinsley, S.A., Bellamy, P.E., Enoksson, B., Fry, G., Gabrielsen, L., McCollin, D. & Schotman, A. (1998) Geographical and land-use influences on bird species richness in small woods in agricultural landscapes. *Global Ecology & Biogeography Letters* 7, 125-135.

Hoffman, M.T., Todd, S.W., Ntshona, Z.M. & Turner, S.D. (1999). Land Degradation in South Africa. National Botanical Institute, Cape Town.

Hull, H.E., Chown, S.L. & Bellamy, C.L. (1998). Identification and evaluation of priority vonservation areas for Buprestidae (Coleoptera) in South Africa, Lesotho, Swaziland and Namibia. *African Entomology* 6, 265-274.

IUCN. (1991). Caring for the earth: a strategy for sustainable living. IUCN UNEP WWF, Gland, Switzerland.

James, V.U. (1991). *Resource Management in Developing Countries: Africa's Ecological and Economic Problems.* pp 13-42 & 79-94. Bergin & Garvey, London.

Joubert, D.F. (1998). *Small mammal and bird community structure in commercial and communal rangelands in a semi-arid shrubland in Namaqualand, South Africa.* Unpublished MSc dissertation: University of Cape Town, Cape Town, South Africa.

Keith, D. (1995). Involving ecologists and local communities in survey, planning and action for conservation in a rural landscape: an example from the Bega Valley, New South Wales. In *Nature Conservation 4 - the Role of Networks* (Eds. D.A. Saunders, J.L. Craig & E.M. Mattiske), pp 385-400. Surrey Beatty, Sydney.

Koch, S.O., Chown, S.L., Davis, A.L.V., Endrödy-Younga, S. & van Jaarsveld, A.S. (2000). Conservation strategies for poorly surveyed taxa: a dung beetle (Coleoptera, Scarabaeidae) case study from southern Africa. *Journal of Insect Conservation* 4, 45-56.

Kunin, W.E. & Lawton, J.H. (1996). Does biodiversity matter? Evaluating the case for conserving species. In *Biodiversity: A biology of numbers and difference* (Ed. KJ Gaston), ch 11. Blackwell Science Ltd, Oxford, UK.

Land Type Survey Staff. (1986). Land types of the map 2730 Vryheid. *Memoirs of Agricultural Natural Resouces of South Africa* No. 7.

Lawes, M.J., Eeley, H.A.C. & Piper, S.E. (*In Press*). The relationship between local and regional diversity of indigenous forest fauna in KwaZulu-Natal Province, South Africa. *Biodiversity and Conservation.*

Lewis, L.A. & Berry, L. (1988). *African Environments and Resources*. Unwin Hyman, London.

Lombard, A.T., Cowling, R.M., Pressey, R.L. & Mustart, P.J. (1997). Reserve selection in a species-rich and fragmented landscape on the Augulhas plain, South Africa. *Conservation Biology* 11, 1101-1116.

Low, A.B. & Rebelo, A.G. (1996). *Vegetation of South Africa, Lesotho and Swaziland.* Department of Environmental Affairs and Tourism, Pretoria.

Macdonald, I.A.W. (1989). Man's role in changing the face of southern Africa. In *Biotic diversity in Southern Africa. Concepts and conservation* (Eds. B.J. Huntley), pp 51-77. Oxford University Press, Cape Town.

MacVicar, C.N., Scotney, D.M., Skinner, T.E., Niehaus, H.S. & Loubser, J.H. (1974). A classification of land (climate, terrain form, soil) primarily for rainfed agriculture. *South African Journal of Agriculture Extensions* 3, 21-24.

Maddock, A. & Du Plessis, M.A. (1999). Can species data only be appropriately used to conserve biodiversity. *Bioiversity & Conservation* 8, 603-615.

Margules, C.R. & Pressey, R.L. (2000). Review article: Systematic conservation planning. *Nature* 405, 243-253.

McCann, K.S. (2000). Review article: Global patterns in biodiversity. *Nature* 405, 220-227.

Milton, S.J. & Macdonald, I.A.W. (1988). Tree deaths near tar roads in the Northern Transvaal. *South African Journal of Science* 84, 164-165.

Mishra, C. & Rawat, G.S. (1998). Livestock grazing and biodiversity conservation: comments on Saberwal. *Conservation Biology* 12, 712-714.

Muller, C. (1999). *The distribution and conservation of termites (Isoptera) and butterflies (Lepidoptera) in South Africa*. Unpublished MSc dissertation: University of Pretoria, Pretoria, South Africa.

Nantel, P., Bouchard, A., Brouillet, L. & Hay, S. (1998). Selection of areas for protecting rare plants with integration of land-use conflicts: A case study for the west coast of Newfoundland, Canada. *Biological Conservation* 84, 223-234.

National Water Act (Act 36 of 1998). Government Gazette No 19182, 26 August 1998. Pretoria.

NDA. (1995). *White paper on agriculture*. National Directorate of Agriculture, Pretoria.

NDA. (2000). *Economic review of the South African Agriculture, 1999/2000*. National Directorate of Agriculture, Pretoria.

Newby, T. & Wessels, K.J. (1997). *Report on the extent of natural resource degradation in South Africa.* Institute for Soil, Climate and Water Report GW/A/97/57, mimeo, Pretoria, South Africa.

Noss, R.F. (1996). Ecosystems as conservation targets. *Trends in Ecology & Evolution.*11, 351.

Pressey, R.L. (1992). Nature conservation in rangelands: Lessons from research on reserve selection in New South Wales. *Rangelands Journal* 14, 214-226.

Pressey, R.L. (1994). Land classifications are necessary for conservation planning but what do they tell us about fauna. In *Future of the Fauna of Western New South Wales*. (Eds. D. Lunney, S. Hand, P. Reed & D. Butcher), ch 3. The Royal Zoological Society of New South Wales, Australia.

Pressey, R.L. (1997). Priority conservation areas: towards an operational definition for regional assessments. In *National parks and protected areas: selection, delimitation, and management* (Eds. J.J. Pigram & R.C. Sundell), pp 337-357. Centre for Water Policy Research, University of New Engalnd, Australia.

Pressey, R.L. (1998). Algorithms, politics and timber: an example of the role of science in a public, political negotiation process over new conservation areas in production forest. In *Ecology for everyone: communicating ecology to scientists, the public and the politicians*. (Eds. R.T. Willis & R.J. Hobbs), pp 73-87. Surrey Beatty & Sons, Sydney.

Pressey, R.L., Ferrier, S., Hager, T.C., Woods, C.A., Tully, S.L. & Weinman, K.M. (1996). How well protected are the forests of north-eastern New South Wales? – analyses of forest environments in relation to formal protection measures, land tenure and vulnerability to clearing. *Forest Ecology and Management* 85, 311-333.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993). Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology & Evolution* 8, 124-128.

Pressey, R.L., Johnson, I.R. & Wilson, P.D. (1994). Shades of irreplaceability: towards a measure of the contribution of sites to a reservation goal. *Biodiversity & Conservation* 3, 242-262.

Pressey, R.L. & Logan, V.S. (1997). Inside looking out: Findings of research on reserve selection relevant to "off-reserve" nature conservation. In *Conservation Outside Nature Reserves*. (Eds. P. Hale & D. Lamb), pp. 407-418. Centre for Conservation Biology, University of Queensland.

Reijen, R., Foppen, R., ter Braak, C. & Thissen, J. (1995). The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to the proximity of main roads. *Journal of Applied Ecology* 32, 187-202.

Richardson, D.M. & van Wilgen, B.W. (1986). Effects of thirty-five years of afforestation with *Pinus radiata* on the composition of of mesic mountain fynbos near Stellenbosch. *South African Journal of Botany* 52, 309-315.

Rodrigues, A.S.L., Gregory, R.D. & Gaston, K.J. (2000). Robustness of reserve selection procedures under temporal species turnover. *Proceedings of the Royal Society, London (B)* 267, 49-55.

Rottenborn, S.C. (1999). Prediciting the impacts of urbanisation on riparian bird communities. *Biological Conservation* 88, 289-299.

SA Surveyor General. (1993). *1:500 000 scale road network.* Department of Land Affairs, Division of Surveying and Mapping, Cape Town, South Africa.

Sala, E.S., Chaplin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D,M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. & Wall, D.H. (2000). Global biodiversity scenarios doe the year 2100. *Science* 287, 1770-1774.

Saunders, D.A., Hobbs, R.J. & Arnold, G.W. (1993). The Kellerberrin project on fragmented landscapes: A review of current information. *Biological Conservation* 64, 185-192.

Saunders, D.A., Hobbs, R.J. & Margules, C.R. (1991). Biological consequences of ecosystem fragmentation: a review. *Consevation Biology* 5, 18-32.

Schoeman, J.L. & Scotney, D.M. (1987). Agricultural potential as determined by soil, terrain and climate. *South African Journal of Science* 83, 260-268.

Schoeman, J.L., Ellis, F., Turner, D.P., MacVicar, C.N., Hartmann, M.O. & Scotney, D.M. (1986). *An overview of crop production potential and erodibility of land in the RSA*. Map Number GB/B/86/12441. Institute for Soil, Climate and Water, Pretoria.

Scholtz, C.H. & Chown, S.L. (1993). Insect conservation and extensive agriculture: the savanna of southern Africa. In *Perspectives on Insect Conservation* (Eds. K.J. Gaston, T.R. New, & M.J. Samways), pp 75-95. Intercept, Andover.

Schulze, R.E. (1998). *South African Atlas of Agrohydrology and Climatology Water Research.* Commission report TT82/96, Pretoria, South Africa.

Scotney, D.M., Botha, A.D.P., Cloete, J.G., Cowan, G., Dreyer, L., Koch, F.G.L., LeRoux, G.H., Little, A.M., Olivier, J. & Saunders, J. (1988). Agricultural areas of southern Africa. In *Long-term data series relating to southern Africa's renewable natural resources*. (Eds. I.A.W. Macdonald & R.J.M. Crawford), South African National Scientific Programmes Report no. 157, 316-336. Pretoria: CSRI.

Seymour, C. (1998). *Different grazing intensities in arid rangelands: Effects on invertebrates on a communal farm in the succulent Karoo, South Africa*. Unpublished MSc dissertation. University of Cape Town, Cape Town, South Africa.

Smith, H.J. (1998). *Definition and demarcation of high potential land for rain-fed annual crop production.* Unpublished M Inst. Agrar. dissertation. University of. Pretoria, Pretoria, South Africa.

Soulé, M.S. (1991). Conservation: Tactics for a Constant Crisis. *Science* 253, 744-750.

Stoms, D.M. (2000). GAP management status and regional indicators of threats to biodiversity. *Landscape Ecology* 15, 21-33.

Suárez, F., Martinez, C., Herranz, J. & Yanes, M. (1997). Conservation status and farmland requirements of Pintailed Sandgrouse *Pterocles alchata* and Black-bellied Sandgrouse *Pterocles orientalis* in Spain. *Biological Conservation* 82, 73-80.

Thompson, M. (1995). *Update of National Forestry Coverage Map.* Forestek, CSIR, Pretoria, Report: FOR-DEA.

Thompson, M. (1996). The standard land-cover classification scheme for remote-sensing application in South Africa. *South African Journal of Science* 92, 34-42.

Trombulak, S.C. & Frissell, C.A. (2000). Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14, 18-30.

van Jaarsveld, A.S., Ferguson, J.W.H. & Bredenkamp, G.J. (1998a). The Groenvaly grassland fragmentation experiment: design and initiation. *Agricultural Ecosystems & Environments* 68, 139-150.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998b). Biodiversity assessments and conservation strategies. *Science* 279, 2106-2108.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. & Melillo, J.M. (1997). Human domination of Earth's ecosystems. *Science* 277, 494-499.

WCED (World Commission on Environment and Development; The Brundtland Commission). (1987). *Our Common Future.* Oxford University Press. Oxford, UK.

White, D., Minotti, P.G., Barczak, M.J., Sifneos, J.C., Freemark, K.E., Santelmann, M.V., Steinitz, C.F., Kiester, A.R. & Preston, E.M. (1997). Assessing risk to Biodiversity from future landscape change. *Conservation Biology* 11, 349-360.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C. & Pressey, R. (1996). A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. *Conservation Biology* 10, 155-174.

Williams, P.H. (1998). Key sites for conservation: area selection methods for biodiversity. In Conservation in a Changing World: Integrating Processes into Priorities for Action. (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg), pp 211-249. Cambridge University Press, UK.

Wilson, M.G.C. & Anhaeusser, C.R. (Eds.) (1998). *The mineral resources of South Africa: Handbook*, Council for Geoscience, 16, 740pp.

**Appendix 1:** Percentage area currently transformed, suitable for alternate land-uses and impacted on by road-effects within each of the grid cells selected by the combined algorithm.

| Grid cell | Transformed | Suitable | Cultivated | Rainfed | Irrigated | Mining | Dimension | Mineral | Road-effect | Forest | Eucalyptus | E.c. | E.t. | E.u. | E.n. | E.s. | Pine | P.c. | P.t. | P.p | Wattle |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 11 | 7 | 7 | 5 | 0 | 5 | 0 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12 | 2 | 13 | 3 | 0 | 3 | 1 | 0 | 1 | 4 | 5 | 5 | 4 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 13 | 3 | 4 | 0 | 0 | 0 | 2 | 0 | 1 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 20 | 7 | 47 | 0 | 0 | 0 | 2 | 0 | 2 | 4 | 44 | 44 | 43 | 16 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 24 | 4 | 49 | 7 | 1 | 6 | 1 | 1 | 0 | 5 | 40 | 40 | 33 | 21 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| 25 | 10 | 73 | 4 | 4 | 0 | 1 | 0 | 1 | 13 | 70 | 70 | 52 | 44 | 5 | 0 | 8 | 0 | 0 | 0 | 0 | 12 |
| 27 | 4 | 8 | 1 | 0 | 1 | 5 | 4 | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 29 | 11 | 7 | 4 | 0 | 4 | 1 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 32 | 29 | 87 | 14 | 14 | 0 | 4 | 0 | 4 | 3 | 86 | 86 | 47 | 59 | 33 | 3 | 36 | 19 | 19 | 2 | 5 | 23 |
| 33 | 49 | 98 | 33 | 33 | 0 | 1 | 0 | 1 | 3 | 97 | 97 | 29 | 63 | 68 | 0 | 69 | 55 | 55 | 37 | 7 | 16 |
| 34 | 49 | 76 | 7 | 0 | 7 | 3 | 0 | 3 | 2 | 68 | 68 | 59 | 38 | 7 | 0 | 8 | 1 | 1 | 0 | 0 | 0 |
| 39 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 52 | 9 | 39 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 39 | 39 | 39 | 36 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 58 | 2 | 100 | 1 | 0 | 1 | 1 | 0 | 1 | 3 | 99 | 99 | 99 | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 67 | 66 | 91 | 0 | 0 | 0 | 1 | 0 | 1 | 2 | 90 | 90 | 90 | 29 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 69 | 13 | 79 | 0 | 0 | 0 | 2 | 0 | 2 | 5 | 77 | 77 | 77 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 77 | 30 | 52 | 6 | 2 | 4 | 0 | 0 | 0 | 6 | 43 | 43 | 43 | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| 83 | 23 | 81 | 2 | 2 | 0 | 5 | 0 | 4 | 13 | 75 | 75 | 72 | 29 | 2 | 0 | 3 | 0 | 0 | 0 | 0 | 3 |
| 87 | 47 | 79 | 0 | 0 | 0 | 0 | 0 | 0 | 14 | 75 | 75 | 75 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 89 | 27 | 73 | 15 | 15 | 0 | 0 | 0 | 0 | 6 | 68 | 68 | 58 | 36 | 6 | 0 | 8 | 1 | 1 | 0 | 0 | 6 |
| 91 | 56 | 95 | 48 | 48 | 0 | 1 | 0 | 1 | 9 | 94 | 94 | 47 | 21 | 22 | 39 | 45 | 35 | 34 | 24 | 28 | 53 |

| | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 92 | 42 | 100 | 36 | 35 | 1 | 1 | 0 | 1 | 6 | 99 | 99 | 51 | 65 | 43 | 2 | 43 | 32 | 32 | 21 | 15 | 4 |
| 95 | 60 | 100 | 6 | 6 | 0 | 2 | 0 | 2 | 2 | 99 | 99 | 99 | 32 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 102 | 62 | 100 | 70 | 70 | 1 | 0 | 0 | 0 | 10 | 99 | 99 | 12 | 56 | 86 | 11 | 86 | 83 | 82 | 68 | 53 | 11 |
| 103 | 69 | 100 | 41 | 24 | 22 | 0 | 0 | 0 | 10 | 99 | 99 | 79 | 77 | 14 | 0 | 15 | 3 | 3 | 0 | 0 | 0 |
| 106 | 10 | 100 | 34 | 24 | 12 | 1 | 1 | 0 | 8 | 99 | 99 | 97 | 43 | 1 | 0 | 2 | 0 | 0 | 0 | 0 | 0 |
| 124 | 9 | 19 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 19 | 19 | 19 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 137 | 3 | 78 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 76 | 72 | 66 | 10 | 0 | 8 | 0 | 0 | 0 | 0 | 0 | 21 |
| 138 | 8 | 98 | 22 | 16 | 6 | 0 | 0 | 0 | 2 | 97 | 97 | 97 | 62 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 141 | 19 | 99 | 19 | 12 | 7 | 4 | 0 | 4 | 6 | 95 | 95 | 95 | 37 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 145 | 17 | 87 | 6 | 6 | 0 | 9 | 0 | 9 | 4 | 84 | 84 | 84 | 55 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 147 | 11 | 97 | 0 | 0 | 0 | 1 | 0 | 1 | 3 | 96 | 96 | 95 | 1 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 |
| 149 | 18 | 94 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 93 | 92 | 77 | 3 | 0 | 27 | 0 | 0 | 0 | 0 | 0 | 8 |
| 152 | 29 | 98 | 10 | 10 | 0 | 1 | 0 | 0 | 6 | 97 | 97 | 92 | 17 | 0 | 8 | 0 | 0 | 0 | 0 | 0 | 11 |
| 153 | 34 | 100 | 26 | 26 | 0 | 4 | 0 | 4 | 13 | 99 | 99 | 99 | 43 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 157 | 52 | 99 | 27 | 27 | 0 | 4 | 0 | 3 | 16 | 97 | 97 | 97 | 49 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 158 | 23 | 99 | 3 | 3 | 0 | 3 | 0 | 2 | 11 | 97 | 97 | 97 | 58 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 162 | 14 | 100 | 11 | 11 | 0 | 2 | 0 | 2 | 12 | 99 | 99 | 98 | 0 | 0 | 14 | 0 | 0 | 0 | 0 | 0 | 3 |
| 163 | 23 | 100 | 1 | 1 | 0 | 5 | 0 | 5 | 13 | 99 | 99 | 99 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 166 | 45 | 100 | 11 | 11 | 0 | 1 | 0 | 1 | 3 | 98 | 98 | 97 | 4 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 3 |
| 177 | 67 | 86 | 0 | 0 | 0 | 1 | 0 | 1 | 2 | 84 | 84 | 84 | 1 | 0 | 28 | 0 | 0 | 0 | 0 | 0 | 0 |
| 178 | 5 | 100 | 9 | 9 | 0 | 2 | 0 | 2 | 1 | 99 | 98 | 59 | 63 | 23 | 5 | 34 | 18 | 17 | 3 | 6 | 45 |
| 179 | 52 | 100 | 30 | 30 | 0 | 6 | 0 | 6 | 7 | 99 | 99 | 73 | 75 | 18 | 0 | 20 | 16 | 16 | 1 | 2 | 11 |
| 181 | 9 | 100 | 6 | 6 | 0 | 0 | 0 | 0 | 1 | 99 | 99 | 53 | 72 | 25 | 0 | 29 | 19 | 18 | 1 | 11 | 24 |
| 185 | 24 | 81 | 14 | 0 | 14 | 1 | 0 | 1 | 7 | 79 | 79 | 75 | 11 | 3 | 0 | 3 | 1 | 1 | 0 | 0 | 0 |
| 187 | 19 | 42 | 7 | 7 | 0 | 0 | 0 | 0 | 1 | 41 | 41 | 40 | 16 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 3 |
| 188 | 12 | 20 | 3 | 3 | 0 | 1 | 0 | 1 | 0 | 20 | 20 | 19 | 11 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 4 |

| | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 190 | 24 | 25 | 3 | 0 | 3 | 0 | 0 | 0 | 2 | 23 | 23 | 11 | 18 | 11 | 0 | 13 | 6 | 6 | 2 | 0 | 0 |
| 191 | 59 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 2 | 0 | 2 | 2 | 0 | 2 | 2 | 2 | 1 | 0 | 0 |
| 192 | 4 | 36 | 0 | 0 | 0 | 1 | 0 | 1 | 2 | 34 | 34 | 34 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 194 | 5 | 97 | 4 | 0 | 4 | 0 | 0 | 0 | 5 | 96 | 96 | 96 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 204 | 24 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 205 | 58 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 213 | 18 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 8 | 8 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 2 |
| 214 | 55 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 4 | 3 | 4 | 1 | 0 | 1 | 1 | 1 | 0 | 0 | 0 |

*E.c.* = *Eucalyptus camaldulensis*, *E.t.* = *Eucalyptus tereticornis*, *E.u.* = *Eucalyptus urophylla*, *E.n.* = *Eucalyptus nitens*, *E.s.* = *Eucalyptus saligna*.

*P.e.* = *Pinus elliottii*, *P.p.* = *Pinus patula*, *P.t.* = *Pinus taeda*

# CHAPTER 7

# General Discussion

The need for conservation areas, in which biological diversity can be protected from external anthropogenic threats, is becoming increasingly important (Margules & Pressey, 2000). As human populations and their land-use requirements expand, so natural areas in which biodiversity can persist become more threatened. This is of crucial importance not just for the preservation of biodiversity, but for the continued existence of humankind. Biodiversity provides many goods and services on which humans are directly and indirectly reliant, without which our survival is questionable (Kunin & Lawton, 1996). Protected areas in which biodiversity is conserved already exist. However, these areas are inadequate both in terms of coverage and in their representation of biodiversity. The total global land area within conservation areas is estimated to be approximately 7.9%. In addition to this most of the current protected areas were proclaimed in a primarily *ad hoc* and opportunistic fashion, with little regard for the biological patterns and processes (Pressey *et al.*, 1993). These areas were mainly selected on the basis of tourism potential, scenic values, the presence of endemic disease and the lack of agricultural or forestry potential. The resultant biased representation of regional biodiversity and increased costs of achieving adequate representation have led to a rapid proliferation in techniques for the systematic selection of areas important to biodiversity conservation. These techniques aim to represent maximum biodiversity within minimum land area in a region and are relatively efficient in fulfilling this purpose (Williams, 1998). However, there are several obvious shortcomings in these procedures requiring urgent attention before these techniques can effectively be implemented in real-world conservation planning. This study therefore sets about to identify many of these shortcomings and to address them in an effort to improve conservation planning in the Northern Province of South Africa.

Due to the complexity of biodiversity, a complete inventory of biodiversity is generally unattainable (Prendergast *et al.*, 1993). Thus the first shortcoming identified and assessed deals with incomplete biodiversity databases, finding appropriate surrogate or substitute measures for biodiversity and testing their adequacy in conservation planning. The results illustrate that indicator taxa (taxa with well-known distributions and taxonomy) perform well at representing non-target taxa. However, two problem areas are highlighted: first, these conservation areas based on indicator taxa exclude many rare and endemic species of non-target taxa; and second, the assessment techniques used for testing the validity of indicator taxa as biodiversity surrogates are varied and provide different levels of support. As illustrated in both Chapters 2 and 3 levels of overlap between areas of conservation importance to different taxa may be low but are not an indication of the success with which indicator based conservation areas represent biodiversity. Rather one should look at the number of non-target species captured within these areas as a measure of success. This is in agreement with findings by Reid (1998), Howard *et al.* (1998), Prendergast *et al.* (1993) and Lombard (1995). Thus recommendations include the careful consideration of rare and endemic species, as well as the standardisation of assessment techniques.

The realisation that species are only one level of the biodiversity hierarchy has prompted the use of higher hierarchical levels of broad-scale environmental classes including vegetation and land types (Wessels *et al.* 1999). The use of these forms of data in conservation planning in the Northern Province illustrate that increased success in the representation of regional biodiversity (measured as species diversity) comes at an increased cost to land. The results illustrate that the best approach is a combined one using both environmental surrogates as well as species data. Once again the exclusion of rare and endemic biodiversity features through this surrogate-based approach is highlighted. Finally, in a similar fashion to work by Soulé and Sanjayan (1998) these results refute the recommended 10% protected area coverage, illustrating that this target results in the exclusion of many biodiversity features, particularly rare and endemic ones.

Existing conservation area selection techniques have focussed largely on the representation of biodiversity patterns (alpha diversity) and not on the processes responsible for these patterns or turnover in the patterns (beta diversity) (Rodrigues *et al.* 2000). In addition, not many of the existing techniques include measures of threat into conservation planning (Wessels *et al.* 2000). The study addresses these shortcomings through the inclusion of environmental and species gradients, beta diversity patterns and land-use threats into conservation area selection, making these techniques more useful conservation tools. These improvements in conservation planning unfortunately impose increased land costs, but the use of off-reserve management in the human matrix, rather than formal protection, can alleviate some of these demands. In recognition of the fact that current land-use patterns are not static and will expand, natural areas of high suitability for alternate land-uses (e.g. cultivation, forestry and mining) are identified and applied to conservation planning in the Northern Province.

Finally, all the methods developed in this study are used to identify areas of high importance to biodiversity (areas of high biodiversity value). However, the reality of the situation suggests that not all of these areas will receive immediate conservation attention. Therefore, the final analysis sets out to prioritise these areas of high biodiversity value using threat values of current and future land-use threats in an effort to identify those areas requiring immediate conservation attention.

Although this study goes a long way towards addressing many weaknesses highlighted in conservation planning techniques, there are still several problems encountered within the study that deserve mention. These shortcomings have implications for conservation planning and must be considered before implementation of the techniques in real world conservation planning scenarios. Several of these weaknesses are discussed in the introduction and include the lack of presence/absence species distribution data, selection unit size and the resolution of environmental and biological surrogate data. The ideal form of data for the identification of areas important to biodiversity conservation is presence/absence species distribution data, where all areas in the region of interest have been surveyed for the presence of all species. Obviously these data are very labour intensive to obtain and are subsequently scarce. The only such database for the Northern Province is the Bird Atlas Database (Harrison, 1992). The other databases are presence-only databases, including the mammal and butterfly databases. The major problem with datasets of this kind is the potential for false absences and therefore the possibility that areas of high conservation value may be excluded from conservation areas and areas of low value may be included. Therefore most of the study employs the bird data only, however there are sections that require data on other taxa (e.g. indicator work). This requirement, as well as the fact that the other datasets can still make an important contribution to conservation planning makes the inclusion of the other presence-only databases in parts of the study necessary. This is done, however, with full knowledge of these datasets' shortcomings and any conservation outcomes are treated with caution. Similarly the mapping of the vegetation and landtypes is at a very coarse scale, but once again this is the best data available and to exclude it from conservation planning would have more serious consequences for biodiversity conservation. Thus all of these shortcomings associated with the biological and environmental data were understood, acknowledged and taken note of in any recommendations. But until better databases are available, these data form an essential, albeit flawed, component of conservation area selection.

The selection units employed within the study are quarter degree grid squares (QDS's) with an average size of 600-700km$^2$. This is a large size for conservation planning, as this area can contain a multitude of different habitats and species within one grid square. To treat this then as one homogenous unit is very simplistic and misses out on a lot of heterogeneity. In addition many conservation areas are smaller than this planning unit size. Because of the heterogeneity present within the grid cell one cannot assume that a conservation area placed anywhere within that cell will capture and protect all biodiversity found within it. Therefore although these units are useful for assessing some of the questions posed in the study, they are not realistic planning units for conservation. Once again the limitations of the data available imply that we either have to work with the data available or sit back and wait for better data to become available. The latter option seems unadvisable considering the plight that much of biodiversity is in at the moment. One potential solution to this problem of the planning units (QDS's) is to realise the problems associated, the limitations this places on any conservation outputs, and investigate the areas identified at this scale at a more local scale (Wessels *et al.*, 2000). This was the approach taken by the study. Grid cells were used to identify areas of biodiversity conservation importance within the Northern Province. This set of grids is however not a final output of the conservation area selection procedure, it highlights areas which must then be investigated at the local scale in order to identify regions within the QDS's where conservation or off-reserve management in essential.

In conclusion, although the field of conservation planning is beset by weaknesses and inadequacies, it is still an essential component of effective biodiversity conservation. This thesis has succeeded in addressing many of these shortcomings, thereby contributing towards these techniques becoming real-world conservation tools. There are however still many problems with the techniques outlined above. This does not invalidate the techniques, it merely argues a degree of caution in the implementation of the techniques and requires additional local scale work. This does, however, illustrate that there is still much work to be done in the field of conservation planning, from the collection of data all the way to the implementation and management of the area selected. In the South African context, with shortages of conservation resources and funds, as well as land redistribution issues, conservation planning faces many difficulties. Therefore, the need to make these procedures as flexible, efficient, transparent and realistic as possible is essential. The role of off-reserve conservation areas is one that should also be investigated as a potential means for addressing these difficulties and ensuring the persistence of biodiversity in one of the world's most biodiverse regions.

**References**

Harrison, J.A. (1992). The South African Bird Atlas Project: Five years of growth. *South African Journal of Science* 88, 410-413.

Howard, P.C., Viskanic, P., Davenport, T.R.B., Kigenyi, F.W., Baltzer, M., Dickinson, C.J., Lwanga, J.S., Matthews, R.A. & Balmford, A. (1998). Complementarity and the use of indicator groups for reserve selection in Uganda. *Nature* 394, 472-475.

Kunin, W.E. & Lawton, J.H. (1996). Does biodiversity matter? Evaluating the case for conserving species. In *Biodiversity: A biology of numbers and difference.* (Ed. KJ Gaston), ch 11. Blackwell Science Ltd, Oxford, UK.

Lombard, A.T. (1995b). The problems with multi-species conservation: do hotspots, ideal reserves and existing reserves coincide? *South African Journal of Zoology* 30, 145-163.

Margules, C.R. & Pressey, R.L. (2000). Review article: Systematic conservation planning. *Nature* 405, 243-253.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. (1993). Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature* 356, 335-337.

Pressey, R.L., Humphries, C.L., Margules, C.R., Vane-Wright, R.I., & Williams, P.H. (1993). Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology & Evolution* 8, 124-128.

Reid, W.V. (1998). Biodiversity hotspots. *Trends in Ecology & Evolution* 13, 275-280.

Rodrigues, A.S.L., Gregory, R.D. & Gaston, K.J. (2000). Robustness of reserve selection procedures under temporal species turnover. *Proceedings of the Royal Society, London (B)* 267, 49-55.

Soulé, M.E. & Sanjayan, M.A. (1998). Conservation targets: do they help? *Science* 279, 2060-2061.

Wessels, K.J., Freitag, S. & van Jaarsveld, A.S. (1999). The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological Conservation* 89, 21-38.

Wessels, K.J., Reyers, B., Krüger, M. & van Jaarsveld, A.S. (2000). Incorporating land-cover information into regional biodiversity assessments in South Africa. *Animal Conservation* 3, 67-79.

Williams, P.H. (1998). Key sites for conservation: area selection methods for biodiversity. In *Conservation in a Changing World.* (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg), pp 211-249. Cambridge University Press, UK.

# ADDENDUM I

## Priority areas for conserving South African vegetation: a coarse-filter approach

# Priority areas for conserving South African vegetation: a coarse-filter approach

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**Keywords:** Coarse-filter, biodiversity conservation, land-cover, vegetation types, road-effects

**Running title:** South African conservation areas

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**Abstract**

South Africa has an important responsibility to global biodiversity conservation, but a largely inadequate conservation area network for addressing this responsibility. This study employs a coarse-filter approach based on 68 potential vegetation units to identify areas that are largely transformed, degraded or impacted on by road-effects. The assessment highlights broad vegetation types that face high biodiversity losses currently or in the near future due to human impacts. Most vegetation types contain large tracts of natural vegetation, with little degradation, transformation or impacts from road networks. Regions in the grasslands, fynbos and forest biomes are worst affected. Very few of the vegetation types are adequately protected according to the IUCN's 10% protected area conservation target, with the fynbos and savanna biomes containing a few vegetation types that do achieve this arbitrary goal. This investigation identifies areas where limited conservation resources should be concentrated by identifying vegetation types with high levels of anthropogenic land use threats and associated current and potential biodiversity loss.

**Keywords:** Coarse-filter, biodiversity conservation, land-cover, vegetation types, road-effects

**Running title:** South African conservation areas

## Introduction

South Africa contains a wealth of biodiversity within its borders, unequalled by other temperate regions, earning a place in the top 25 most biodiverse nations (WCMC, 1992; Conservation International, 1998). In addition South Africa harbours the fifth highest number of plant species in the world, with the Cape Floristic Region being recognised as one of the six floral kingdoms of the world. This region contains 8200 plant species of which 5682 are endemic and has lost approximately 30.3% of its primary vegetation (Fairbanks *et al.*, 2000; Myers *et al.*, 2000).

Although its responsibility towards global biodiversity conservation is large, South Africa with only 4.8% (DEAT, 1996) (Figure 1a) of its land surface under formal protection falls far short of the IUCN's nominal recommendation of 10% protected area coverage. This coverage also lags behind the 10% average attained by the rest of sub-Saharan Africa, with Botswana reaching 18.5%, Mozambique 12.7% and Namibia 12.4% (WRI, 1994; McNeely 1994; Siegfried et al., 1998). A moderately expanding human population (Central Statistical Survey, 1998) and associated land transformation in South Africa (mainly urbanisation, cultivation and afforestation (Hoffmann, 1997)) leaves 79% of the country covered with natural woody and grassland vegetation communities (Figure 1b) (Fairbanks *et al.*, 2000). Waterbodies and wetlands cover less than one percent of the land surface area, with human land uses making up the remaining 20% (Fairbanks *et al.* 2000). Fairbanks *et al.* (2000) demonstrate that along with the approximately 30% transformation in the fynbos biome, the savanna and grassland biomes are about 10% and 26% transformed and degraded by human land uses respectively (Figure 1c) (see also Thompson *et al.*, In Review). In addition to this there are a total of 1176 species presently recognised as threatened (WRI, 1994; van Jaarsveld, 2000). Thus with these valuable and often endemic biodiversity resources facing ever-increasing threats from human-induced land transformation, and mostly inadequate conservation efforts to stem these threats, South Africa has an obvious responsibility to do more towards the conservation of biodiversity (van Jaarsveld, 2000).

Most of South Africa's existing protected areas were proclaimed in an *ad hoc* fashion, usually because they contained areas with high scenic or tourism potential, contained endemic diseases and did not conflict with other forms of land use (Pringle, 1982; Freitag *et al.*, 1996; Pressey *et al.*, 1993). Because this form of land allocation to conservation is highly inefficient and fails to effectively conserve biodiversity, several techniques have been developed for the systematic selection of land with a high conservation value, i.e. with high levels of biodiversity and large anthropogenic threats facing that biodiversity (for reviews see Williams, 1998; Margules & Pressey, 2000). However, these techniques require data on the distribution of biodiversity and threats facing biodiversity in order to identify areas important to conservation. Because the biodiversity of a region can never be fully observed and inventoried, species distribution data are often used as a surrogate or substitute measure of biodiversity. This form of data however, has a large number of shortcomings associated with it.

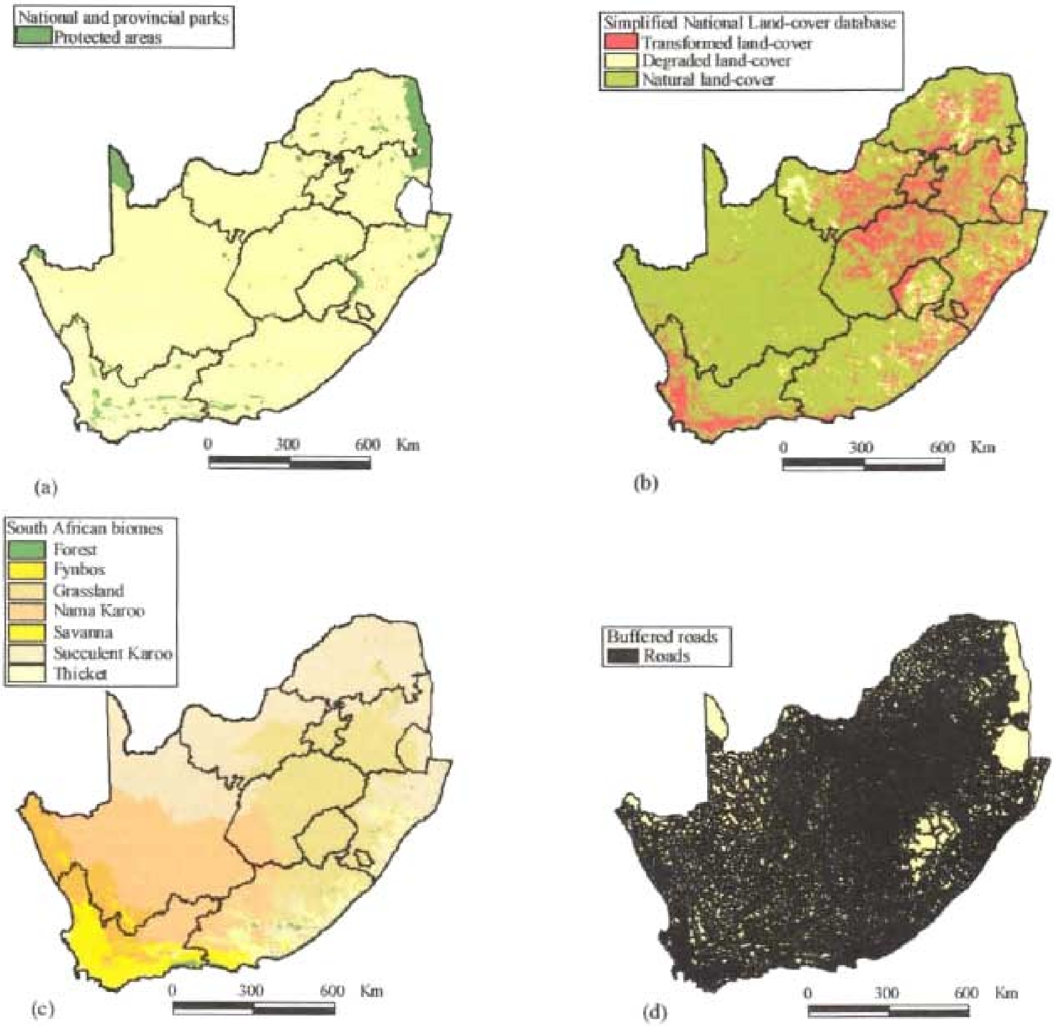


**Figure 1:** Maps of: (a) South African national and provincial protected areas (DEAT, 1996); (b) transformed, degraded and natural land-cover; (c) biomes (Low & Rebelo, 1996); and (d) road network buffered according to Stoms (2000).

These include inadequate taxonomical knowledge of the groups employed, biased sampling efforts and lack of spatial congruency between areas of conservation importance to different taxa (van Jaarsveld *et al*, 1998; Maddock & du Plessis, 1999, Fairbanks & Benn, 2000; Reyers *et al.*, 2000).

*Broad-scale biodiversity surrogates*

In recent years, the focus for conservation has shifted, with recommendations towards a more holistic approach of protecting biodiversity in the aggregate, the so-called 'coarse-filter' approach (Noss, 1987; Noss, 1990). This approach focuses on protecting higher levels of the biodiversity hierarchy (e.g. landclasses and landtypes) rather than species, assuming that these broad-scale biodiversity surrogates represent the finer scale aspects of biodiversity (Williams & Humphries, 1996; Pressey, 1994; Pressey & Logan, 1994; Wessels *et al.*, 1999; Fairbanks & Benn, 2000). However, as Pressey (1994) points out, the assumed relationship between environmental classes and species distribution and abundance is unclear and seldom investigated. In addition, certain species, especially rare species confined to small patches of habitat which are not recognised as distinct environmental classes, may "fall through the coarse filter" when using broad-scale environmental classes (Noss, 1983; Bedward, 1992; Panzer & Schwartz, 1998). Despite the shortcomings associated with a species-based approach to conservation planning, these higher order biodiversity surrogates may well fail to identify the composition, configuration and quantity of elements necessary for biodiversity retention, making species data a necessary component of the conservation planning process (Lambeck, 1997). The shortcomings of species distribution data and the limitations of environmental surrogate measures in the selection of priority conservation areas suggest that perhaps a combination of the two approaches in conservation planning may be advisable (Maddock & du Plessis, 1999).

At a national scale South Africa has a few databases of broader surrogates for biodiversity, including Acocks' Veld Types (Acocks, 1988) and the more recent Vegetation of South Africa, Lesotho and Swaziland (Low & Rebelo, 1996; McDonald, 1997). Acocks (1988) defined biological resources from a purely agricultural potential perspective, while Low and Rebelo (1996) looked at the definition of these resources from a management and potential use angle. These vegetation units were defined as having, "... similar vegetation structure, sharing important plant species, and having similar ecological processes." Thus, these are units that would have potentially occurred today, were it not for all the major human-made transformations e.g. agriculture and urbanisation. Therefore the Low and Rebelo (1996) vegetation map contains significant potential for acting as a broad scale surrogate of South African biodiversity and for identifying land important to biodiversity conservation.

## Methods

*Current land-cover data*

Before the Low and Rebelo (1996) map can be used one has to differentiate between the potential vegetation cover of regions (as defined by Low & Rebelo, 1996) and that which is in reality found in the region. In other words one needs an indication of current natural vegetation pattern, degree of transformation, and amount of protection afforded each vegetation type before one can decide if it constitutes a conservation priority (Rebelo, 1997). As Low and Rebelo (1996) point out "there is little point in setting aside more of a vegetation type with vast expanses in pristine condition, while ignoring the last patches of a type which is not yet conserved." Low and Rebelo (1996) provide some estimates of protection and transformation data, however as they admit, "these are woefully incomplete". Thus, some indication of current land-cover (the suite of natural and human-made features that cover the earth's immediate surface) at a national scale is required for effective land-use planning, sustainable resource management, environmental research and in this instance conservation planning (Rebelo, 1997; Fairbanks *et al.*, 2000).

To this end the advent of the National Land-cover (NLC) database is of extreme relevance. This national database was derived using manual photo-interpretation techniques from a series of 1:250,000 scale geo-rectified hardcopy satellite imagery maps, based on seasonally standardised, single date Landsat Thematic Mapper (TM) satellite imagery captured principally during the period 1994-95 (Fairbanks & Thompson, 1996). It provides the first single standardised database of current land-cover information for the whole of South Africa, Lesotho and Swaziland (Fairbanks *et al.*, 2000). For the purpose of the present study the 31 land-cover classes were reclassified into three categories: natural, degraded and transformed land-cover (Table 1). Natural land-cover included all untransformed vegetation, e.g. forest, woodland, thicket and grassland. The degraded land-cover category was dominated by degraded classes of land-cover. These areas have a very low vegetation cover in comparison with the surrounding natural vegetation cover and were typically associated with rural population centres and subsistence level farming, where fuel-wood removal, over-grazing and subsequent soil erosion were excessive (Thompson 1996). The transformed category consisted of areas where the structure and species composition were completely or almost completely altered which includes all areas under crop cultivation, forestry plantations, urbanised areas, and mines/quarries.

The databases of potential vegetation cover and current land-cover were overlaid in a geographic information system (GIS) to determine the extent of natural, degraded and transformed area within each of the 68 vegetation types identified in Low and Rebelo (1996). These values could then be used to highlight areas of high current and future vulnerability to biodiversity loss through land use impacts. Levels of transformation were compared against the transformation thresholds predicted by a geometric model developed by Franklin and Forman (1987). This work suggested that the most critical time for land planning and conservation is when between 10-40% of the landscape has been transformed or impacted upon. Specifically, most of the rapid ecological changes (e.g., loss of interior species) can be expected when this level increases from 20-40%. Regions showing greater than 40% loss of natural habitat have already undergone significant ecological disruptions.

**Table 1**: Land-cover classes reclassified into broad categories

| Transformation category | % area | Land-cover class |
|---|---|---|
| **Natural land-cover** | 73.4% | Wetlands, grassland, shrubland, bushland, thicket, woodland, forest |
| Degraded land-cover | 10.1% | Degraded land, erosion scars, waterbodies |
| **Transformed land-cover** | 16.5% | Cultivated lands, urban/built-up areas, mines and quarries, forestry plantations |

An additional GIS layer of protected area coverage for the country (DEAT, 1996) was also employed to determine the extent of conservation areas existing within the vegetation types.

*Patterns of roads*

In addition to these land use threats, one of the most widespread forms of alteration of natural habitats and landscapes over the last century has been the construction and maintenance of roads (Trombulak & Frissell, 2000). Road networks affect landscapes and biodiversity in seven general ways: (1) increased mortality from road construction; (2) increased mortality from vehicle collisions; (3) animal behaviour modification; (4) alteration of the physical environment; (5) alteration of the chemical environment; 6) spread of exotic species, and (7) increased alteration and use of habitats by humans (from Trombulak & Frissell, 2000). These networks cover 0.9% of Britain and 1.0% of the USA (Forman & Alexander, 1998), however the road-effect zone, the area over which significant ecological effects extend outward from the road, is usually much wider than the road and roadside. This road effect zone can thus provide an additional estimate of areas with a high vulnerability to biodiversity loss through changing land uses and increased human impacts.

Some evidence on the size of the road-effect zone is available from studies in Europe and North America. Reijnen *et al.* (1995) estimated that road-effect zones cover between 12-20% of The Netherlands, while Forman (2000) illustrated that 19% of the USA is affected ecologically by roads and associated traffic. The road-effect zone for South Africa was determined using a similar method to that used by Stoms (2000) in which the spatial extent of road effects can be used as an ecological indicator that directly represents impacts on biodiversity. For this, the road-effect zone was used as a measure of the area potentially affected by roads. The affected distances were estimated from the reviews mentioned above, as well as from local studies (Milton & MacDonald, 1988). Therefore national routes and freeways were assumed to affect biodiversity for a greater distance from the roadway (1 km on each side) than farm roads (100 m, Table 2).

Road segments from the South African Surveyor General 1993 1:500,000 scale map series files (SA Surveyor General, 1993) were buffered using a standard GIS operation to the distance related to its class (Figure 1d). Although the roads in protected areas do have an impact on biodiversity within these areas, they were excluded from this analysis as by and large protected areas overwhelmingly contribute to biodiversity conservation. A road-effect zone was calculated for the remaining untransformed areas within each vegetation type by summing the total area within the road effect zone surrounding roads in each vegetation type and converting to a percentage of the total remaining untransformed area in that vegetation type. However, the road-effect zone used here does not consider the spatial pattern of roads. So, although roads clearly have a significant impact on many species, meaningful indicators of road-effects on landscapes await the attention of landscape ecologists and other scientists (Forman, 1998).

**Table 2**: Buffer widths assigned to road classes for calculating road effect zone (after Stoms 2000).

| South African Surveyor General Description | Buffer width (m) |
|---|---|
| **National route** | 1000 |
| Freeway | 1000 |
| **Arterial** | 500 |
| Main | 250 |
| Secondary (connecting and magisterial roads) | 100 |
| Other (rural road) | 50 |
| Vehicular trail (4 wheel drive route) | 25 |

Most of these factors also vary over daily, weekly, and annual cycles, which may interfere with critical behavioural periods such as breeding or migration. As such, the road-effect zone can represent only a first order approximation attempt to capture more of the multi-dimensional nature of road network effects.

**Results and Discussion**

*Vulnerability assessment of vegetation types*

The majority of vegetation types of South Africa are not largely degraded or transformed (Table 3). Of the 68 vegetation types 61 contain more than 50% natural vegetation cover with a median value of 81.1% natural vegetation cover across all vegetation types. The vegetation types show low levels of degradation with a median value of 2.8%, with all but one (Afro Mountain Grassland) being less than 20% degraded (Table 3). Only five of the vegetation types are more than 50% transformed by anthropogenic land uses, with a median of 10% being transformed within vegetation types.

Figure 2 provides a diagrammatic representation of the current levels of transformation, degradation and protection across all vegetation types. Similar to the findings of the coarse-scale species-based approach used by Rebelo (1997), the grasslands and fynbos have experienced the most transformation (see Fairbanks *et al.*, 2000), with the coastal indigenous forests having been subjected to extensive transformation for its size (Figures 2a, b). Although degradation levels are generally low, a few regions in the grasslands biome as well as a few in the savanna biome show the highest levels of degradation ranging from 10 to 36% of the vegetation extent (Figure 2c).

The average amount of vegetation type currently under protection is 9.6% (median value of 1.5%) with only 18 vegetation types conforming to the IUCN's nominal recommendation of 10% protected area coverage (Table 3). However, this well cited protected area recommendation of 10% is widely criticised as too little to guarantee the persistence of biodiversity within the region. Soulé & Sanjayan (1998) illustrate that up to 50% of land area may be required to successfully represent all biodiversity elements. Therefore, perhaps even these 18 supposedly well-protected vegetation types are inadequately protected (Figure 2d).

The road-effect zone impacts on an average of 5.5% (with a median value of 6) of the remaining natural land-cover in all vegetation types (Table 3). Five vegetation types (Mesic Succulent Thicket, Moist Clay Highveld Grassland, Dune Thicket, Eastern Thorn Bushveld, Rocky Highveld Grassland) containing between 10 and 14.2% road-effect zones (Table 3). The rest of the vegetation types lie under this 10% level, with the Mopane Shrubveld containing no road-effect due to the fact that it all falls entirely within the boundaries of the Kruger National Park (Table 3).

In Table 4 the areas within each vegetation type that are transformed, degraded or exposed to road-effects are summed to provide an indication of vegetation that has been disturbed or affected by these human land uses.

**Table 3:** Percentage natural, degraded, transformed and protected area of each of the vegetation types, as well as the percentage of each vegetation type exposed to road-effect zones.

| Code | Vegetation type | % natural | % degraded | % transformed | % protected | % road-effect |
|---|---|---|---|---|---|---|
| 1 | Coastal Forest | 89.3 | 1.2 | 9.3 (43) | 1.3 (9.5) | 6.5 |
| 2 | Afromontane Forest | 67.9 | 2.9 | 29.2 (44) | **16.1** (17.6) | 6.4 |
| 3 | Sand Forest | 72.3 | 15.6 | 5.8 (45) | **46.7** (44.6) | 1.7 |
| 4 | Dune Thicket | 62.2 | 8.5 | 27.6 (25) | **10.6** (14.5) | 11.2 |
| 5 | Valley Thicket | 72.1 | 13.0 | 14.8 (51) | 1.5 (2.1) | 6.1 |
| 6 | Xeric Succulent Thicket | 95.0 | 2.0 | 3.0 (51) | 4.6 (8.0) | 6.4 |
| 7 | Mesic Succulent Thicket | 78.5 | 7.0 | 14.5 (51) | 4.0 (5.3) | 14.2 |
| 8 | Spekboom Succulent Thicket | 93.1 | 4.2 | 2.6 (unknown) | 1.2 (1.8) | 4.9 |
| 9 | Mopane Shrubveld | 100.0 | 0.0 | 0.0 (0) | **100** (100) | 0.0 |
| 10 | Mopane Bushveld | 92.4 | 0.9 | 6.6 (8) | **34.0** (38.3) | 3.0 |
| 11 | Soutpansberg Arid Mountain Bushveld | 83.8 | 10.2 | 6.0 (65) | **10.1** (12.6) | 4.3 |
| 12 | Waterberg Moist Mountain Bushveld | 90.2 | 0.8 | 9.0 (28) | 6.2 (8.6) | 3.2 |
| 13 | Lebombo Arid Mountain Bushveld | 90.2 | 0.1 | 9.1 (unknown) | **37.1** (38.0) | 1.0 |
| 14 | Clay Thorn Bushveld | 58.7 | 7.1 | 34.1 (60) | 1.0 (0.9) | 5.1 |
| 15 | Subarid Thorn Bushveld | 78.7 | 12.6 | 8.7 (unknown) | 0.0 (0.2) | 8.2 |
| 16 | Eastern Thorn Bushveld | 69.7 | 13.8 | 16.5 (unknown) | 0.2 (0.5) | 11.1 |
| 17 | Sweet Bushveld | 78.3 | 12.0 | 9.5 (27) | 1.8 (2.3) | 4.5 |
| 18 | Mixed Bushveld | 69.3 | 14.1 | 16.6 (60) | 3.6 (3.1) | 5.3 |
| 19 | Mixed Lowveld Bushveld | 70.4 | 9.9 | 19.8 (30) | **22.5** (28.3) | 3.1 |
| 20 | Sweet Lowveld Bushveld | 85.1 | 1.4 | 13.5 (30) | **62.2** (67.3) | 1.1 |
| 21 | Sour Lowveld Bushveld | 54.4 | 9.6 | 36.0 (76) | 7.0 (9.7) | 4.7 |
| 22 | Subhumid Lowveld Bushveld | 84.1 | 12.3 | 3.6 (36) | **20.9** (21.5) | 1.1 |
| 23 | Coastal Bushveld-Grassland | 43.5 | 15.9 | 39.8 (unknown) | **13.5** (14.0) | 5.9 |
| 24 | Coast-Hinterland Bushveld | 56.7 | 8.2 | 35.0 (87) | 2.1 (3.6) | 4.4 |
| 25 | Natal Central Bushveld | 72.2 | 9.9 | 18.0 (80) | 1.3 (1.6) | 7.2 |
| 26 | Natal Lowveld Bushveld | 72.5 | 11.9 | 15.6 (35) | **14.1** (17.8) | 5.3 |
| 27 | Thorny Kalahari Dune Bushveld | 83.5 | 0.0 | 0.0 (unknown) | **99.6** (99.8) | 0.0 |
| 28 | Shrubby Kalahari Dune Bushveld | 96.0 | 3.1 | 0.0 (55) | **19.4** (19.5) | 2.2 |
| 29 | Karroid Kalahari Bushveld | 98.8 | 1.2 | 0.0 (55) | 0.1 (0.1) | 3.3 |

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|---|---|---|---|---|---|---|
| 30 | Kalahari Plains Thorn Bushveld | 73.6 | 18.9 | 7.1 (55) | 0.5 (0.5) | 3.9 |
| 31 | Kalahari Mountain Bushveld | 99.5 | 0.2 | 0.3 (25) | 0.0 (0.0) | 4.6 |
| 32 | Kimberley Thorn Bushveld | 76.1 | 4.4 | 19.5 (55) | 1.8 (3.1) | 6.8 |
| 33 | Kalahari Plateau Bushveld | 92.7 | 3.0 | 4.2 (55) | 0.0 (0.0) | 5.5 |
| 34 | Rocky Highveld Grassland | 66.3 | 0.1 | 33.6 (65) | 0.8 (1.4) | 10.2 |
| 35 | Moist Clay Highveld Grassland | 68.2 | 0.4 | 31.4 (79) | 0.0 (0.0) | 11.3 |
| 36 | Dry Clay Highveld Grassland | 34.9 | 0.1 | 65.1 (67) | 0.0 (0.0) | 9.0 |
| 37 | Dry Sandy Highveld Grassland | 63.5 | 0.8 | 35.8 (65) | 0.3 (0.3) | 9.1 |
| 38 | Moist Sandy Highveld Grassland | 67.6 | 0.7 | 31.6 (55) | 0.0 (0.7) | 9.4 |
| 39 | Moist Cool Highveld Grassland | 60.4 | 1.6 | 38.0 (72) | 0.7 (0.3) | 9.6 |
| 40 | Moist Cold Highveld Grassland | 46.8 | 11.3 | 41.8 (70) | 0.8 (0.6) | 6.7 |
| 41 | Wet Cold Highveld Grassland | 88.0 | 2.4 | 9.7 (60) | 9.4 (6.7) | 4.1 |
| 42 | Moist Upland Grassland | 61.4 | 17.0 | 21.6 (60) | 2.3 (2.5) | 5.5 |
| 43 | North-eastern Mountain Grassland | 67.6 | 7.1 | 25.3 (45) | 3.3 (7.4) | 4.8 |
| 44 | South-eastern Mountain Grassland | 94.5 | 4.0 | 1.5 (32) | 0.6 (0.3) | 5.7 |
| 45 | Afro Mountain Grassland | 51.9 | 36.7 | 11.4 (32) | 0.0 (0.0) | 0.8 |
| 46 | Alti Mountain Grassland | 87.5 | 8.8 | 3.6 (32) | **11.7** (12.5) | 1.2 |
| 47 | Short Mistbelt Grassland | 38.5 | 4.6 | 56.9 (89) | 0.9 (2.4) | 7.6 |
| 48 | Coastal Grassland | 81.7 | 5.1 | 12.9 (unknown) | 0.1 (1.1) | 7.0 |
| 49 | Bushmanland Nama Karoo | 99.7 | 0.2 | 0.1 (unknown) | 0.0 (0.0) | 3.4 |
| 50 | Upper Nama Karoo | 99.0 | 0.9 | 0.1 (unknown) | 0.0 (0.0) | 5.8 |
| 51 | Orange River Nama Karoo | 98.1 | 0.1 | 1.6 (unknown) | 0.1 (1.5) | 4.6 |
| 52 | Eastern Mixed Nama Karoo | 94.9 | 1.8 | 3.3 (unknown) | 1.6 (1.1) | 7.4 |
| 53 | Great Nama Karoo | 99.1 | 0.8 | 0.2 (unknown) | 0.7 (0.2) | 5.4 |
| 54 | Central Lower Nama Karoo | 90.2 | 9.0 | 0.8 (unknown) | 0.1 (0.0) | 6.0 |
| 55 | Strandveld Succulent Karoo | 86.3 | 2.0 | 9.5 (24) | 0.4 (0.4) | 4.0 |
| 56 | Upland Succulent Karoo | 97.1 | 0.7 | 1.7 (unknown) | 4.2 (4.4) | 4.4 |
| 57 | Lowland Succulent Karoo | 94.2 | 2.6 | 3.2 (unknown) | 0.9 (1.3) | 3.9 |
| 58 | Little Succulent Karoo | 89.0 | 2.6 | 8.4 (unknown) | 3.2 (2.3) | 7.7 |
| 59 | North-western Mountain Renosterveld | 94.0 | 0.0 | 6.0 (unknown) | 0.0 (0.0) | 3.0 |
| 60 | Escarpment Mountain Renosterveld | 98.9 | 0.3 | 0.8 (unknown) | 0.0 (0.1) | 2.4 |
| 61 | Central Mountain Renosterveld | 80.4 | 1.8 | 17.8 (11) | 5.1 (3.6) | 5.4 |
| 62 | West Coast Renosterveld | 9.0 | 1.1 | 89.8 (97) | 0.7 (1.8) | 8.1 |
| 63 | South & South-west Coast Renosterveld | 39.4 | 1.9 | 58.7 (32) | 1.5 (1.4) | 8.8 |
| 64 | Mountain Fynbos | 88.5 | 0.7 | 10.8 (11) | **26.4** (26.1) | 2.9 |
| 65 | Grassy Fynbos | 88.7 | 0.8 | 10.3 (3) | **15.5** (16.1) | 6.0 |

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|---|---|---|---|---|---|---|
| 66 | Laterite Fynbos | 64.8 | 1.1 | 34.1 (50) | 0.0 (0.5) | 8.6 |
| 67 | Limestone Fynbos | 87.2 | 7.6 | 5.2 (40) | **13.6** (13.8) | 4.0 |
| 68 | Sand Plain Fynbos | 34.4 | 8.5 | 57.1 (50) | 1.2 (1.1) | 7.1 |

*(Values in brackets indicate estimates from Low and Rebelo (1996))*

*(Vegetation types with more than 10% protected area coverage are indicated in bold)*

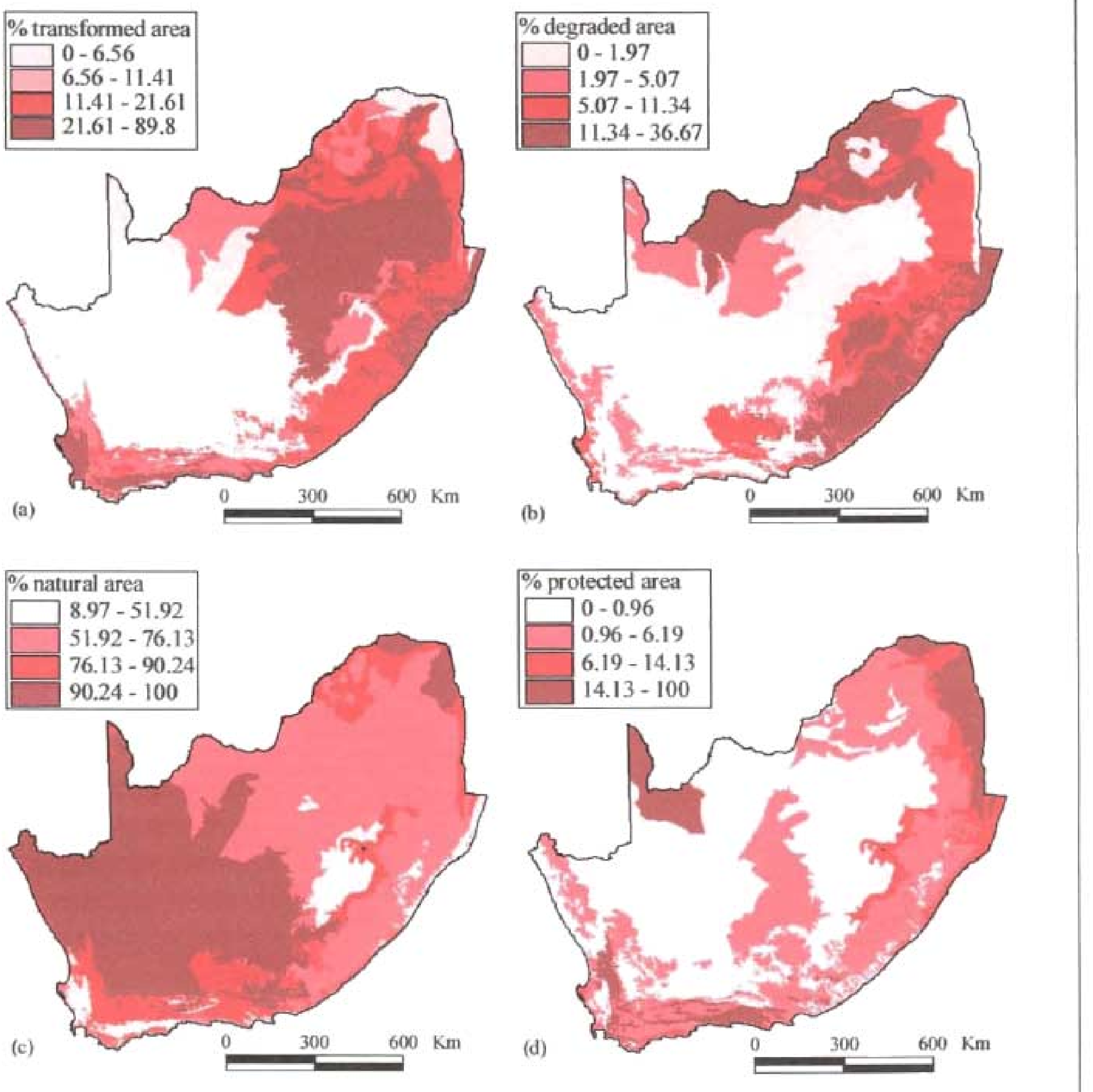


**Figure 2:** Diagrammatic representation of levels of percentage (a) transformed, (b) degraded, (c) natural and (d) protected vegetation cover within each of Low and Rebelo's (1996) vegetation types.

Table 4 provides a list of vegetation types ordered according to their area affected as an indication of their vulnerability to biodiversity loss. Types with large areas affected face a high risk of biodiversity loss due to a combination of extensively degraded and transformed areas with a large road network. The West Coast Renosterveld, Sand Plain Fynbos, Dry Clay Highveld Grassland, South and South-west Renosterveld, Short Mistbelt Grassland, Coastal Bushveld-Grassland, Moist Cold Highveld Grassland, Sour Lowveld Bushveld, Afro Mountain Grassland, Coast-Hinterland Bushveld, Moist Cool Highveld Grassland, Clay Thorn Bushveld, Dune Thicket, Moist Upland Grassland, Dry Sandy Highveld Grassland, Rocky Highveld Grassland and Laterite Fynbos are all areas of concern due to the fact that over 40% of their extent is impacted on by land use threats. This level of land use impact corresponds with the threshold determined by Franklin and Forman (1987), indicating extreme ecological disruption within these vegetation types.

All of these vegetation types are also poorly protected (Table 3) with the Coastal Bushveld-Grassland and Dune Thicket being the only types to reach the IUCN's recommended 10% protected area coverage. However, as stated previously this level of protection is inadequate, especially in the case of these two vegetation types where it would not be sufficient to stem the biodiversity loss associated with such high levels of land use change. Of the 68 vegetation types 38 (56%) fall within the 10-40% category of land use impact determined by Franklin and Forman (1987) and are thus at a critical time for land use planning and conservation.

Table 5 provides a list of the land-cover types within each of the top 10 priority conservation vegetation types drawn from Table 4. The Afro Mountain Grassland and Moist Cold Highveld Grassland contain large areas of degraded vegetation. These same vegetation types along with the West Coast Renosterveld, Sand Plain Fynbos, Dry Clay Highveld Grassland, South and South-west Coast Renosterveld, Short Mistbelt Grassland, Coastal Bushveld-Grassland, Sour Lowveld Bushveld and Coast-Hinterland Bushveld contain extensive areas of commercial, semi-commercial and subsistence dryland cultivation (Table 5). The Short Mistbelt Grassland, Coastal Bushveld-Grassland, Sour Lowveld Bushveld and Coast-Hinterland Bushveld contain large areas of exotic forestry plantations and, with the exception of the Sour Lowveld Bushveld, commercial sugarcane cultivation (Table 5).

Of all these priority vegetation types only the Coastal Bushveld-Grassland has more than 10% protected area coverage at 13.5%, but high levels of degradation as well as high levels of transformation still make it an area of concern along its entire latitudinal distribution. The rest of these top 10 priority vegetation types all fall below five percent protected area coverage (Table 3). This land use analysis is an example of a potential management tool for vulnerable areas, and is not limited to these top 10 vegetation types. Other vegetation types, although not as affected as these 10, are nonetheless also impacted on by land use changes and should therefore also be considered and monitored in a conservation plan. Table 5 is an example of what can be done and similar analyses can be performed on all vegetation types in order to investigate the land use impacts and management parameters within each area.

**Table 4**: Percentage area of vegetation type exposed to the combined land-cover threats of degradation, transformation and road effects

| Code | Vegetation type | Affected area (%) |
|---|---|---|
| 62 | West Coast Renosterveld | 92.3 |
| 68 | Sand Plain Fynbos | 69.5 |
| 36 | Dry Clay Highveld Grassland | 67.8 |
| 63 | South & South-west Coast Renosterveld | 65.4 |
| 47 | Short Mistbelt Grassland | 64.8 |
| 23 | Coastal Bushveld-Grassland | 60.3 |
| 40 | Moist Cold Highveld Grassland | 56.7 |
| 21 | Sour Lowveld Bushveld | 49.1 |
| 45 | Afro Mountain Grassland | 48.6 |
| 24 | Coast-Hinterland Bushveld | 47.0 |
| 39 | Moist Cool Highveld Grassland | 45.8 |
| 14 | Clay Thorn Bushveld | 45.1 |
| 4 | Dune Thicket | 43.9 |
| 42 | Moist Upland Grassland | 42.5 |
| 37 | Dry Sandy Highveld Grassland | 42.3 |
| 34 | Rocky Highveld Grassland | 42.2 |
| 66 | Laterite Fynbos | 40.8 |
| 35 | Moist Clay Highveld Grassland | 39.6 |
| 38 | Moist Sandy Highveld Grassland | 39.3 |
| 16 | Eastern Thorn Bushveld | 38.2 |
| 2 | Afromontane Forest | 37.9 |
| 43 | North-eastern Mountain Grassland | 36.2 |
| 18 | Mixed Bushveld | 34.8 |
| 7 | Mesic Succulent Thicket | 34.0 |
| 25 | Natal Central Bushveld | 33.3 |
| 5 | Valley Thicket | 32.9 |
| 19 | Mixed Lowveld Bushveld | 32.0 |
| 26 | Natal Lowveld Bushveld | 31.6 |
| 32 | Kimberley Thorn Bushveld | 29.4 |

| | | |
|---|---|---|
| 30 | Kalahari Plains Thorn Bushveld | 29.0 |
| 15 | Subarid Thorn Bushveld | 28.0 |
| 61 | Central Mountain Renosterveld | 25.9 |
| 17 | Sweet Bushveld | 25.2 |
| 48 | Coastal Grassland | 23.8 |
| 3 | Sand Forest | 22.8 |
| 11 | Soutpansberg Arid Mountain Bushveld | 20.0 |
| 58 | Little Succulent Karoo | 18.7 |
| 65 | Grassy Fynbos | 17.9 |
| 67 | Limestone Fynbos | 17.2 |
| 22 | Subhumid Lowveld Bushveld | 16.9 |
| 1 | Coastal Forest | 16.8 |
| 41 | Wet Cold Highveld Grassland | 16.2 |
| 20 | Sweet Lowveld Bushveld | 16.0 |
| 54 | Central Lower Nama Karoo | 15.2 |
| 55 | Strandveld Succulent Karoo | 15.1 |
| 64 | Mountain Fynbos | 14.8 |
| 46 | Alti Mountain Grassland | 13.5 |
| 12 | Waterberg Moist Mountain Bushveld | 12.9 |
| 33 | Kalahari Plateau Bushveld | 12.4 |
| 52 | Eastern Mixed Nama Karoo | 12.3 |
| 8 | Spekboom Succulent Thicket | 11.8 |
| 6 | Xeric Succulent Thicket | 11.3 |
| 44 | South-eastern Mountain Grassland | 11.1 |
| 13 | Lebombo Arid Mountain Bushveld | 10.3 |
| 10 | Mopane Bushveld | 10.3 |
| 57 | Lowland Succulent Karoo | 9.5 |
| 59 | North-western Mountain Renosterveld | 9.1 |
| 56 | Upland Succulent Karoo | 6.8 |
| 50 | Upper Nama Karoo | 6.7 |
| 53 | Great Nama Karoo | 6.3 |
| 51 | Orange River Nama Karoo | 6.3 |
| 28 | Shrubby Kalahari Dune Bushveld | 5.2 |
| 31 | Kalahari Mountain Bushveld | 5.1 |
| 29 | Karroid Kalahari Bushveld | 4.5 |
| 49 | Bushmanland Nama Karoo | 3.6 |

| | | |
|---|---|---|
| 60 | Escarpment Mountain Renosterveld | 3.5 |
| 27 | Thorny Kalahari Dune Bushveld | 0.0 |
| 9 | Mopane Shrubveld | 0.0 |

**Table 5:** Description and percentage area coverage of land-cover threats facing conservation priority vegetation types

| Description | West Coast Renosterveld | Sand Plain Fynbos | Dry Clay Highveld Grassland | South & South-west Coast Renosterveld | Short Mistbelt Grassland | Coastal Bushveld-Grassland | Moist Cold Highveld Grassland | Sour Lowveld Bushveld | Afro Mountain Grassland | Coast-Hinterland Bushveld |
|---|---|---|---|---|---|---|---|---|---|---|
| *Natural land-cover* | *9.01* | *34.64* | *34.89* | *39.87* | *39.32* | *43.56* | *46.85* | *54.44* | *51.92* | *56.87* |
| Waterbodies | 0.24 | 0.14 | 0.05 | 0.83 | 0.24 | 4.69 | 0.21 | 0.11 | 0.01 | 0.12 |
| Dongas and sheet erosion scars | 0.00 | 0.05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.09 | 0.00 | 0.00 | 0.00 |
| Degraded: forest and woodland | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.87 | 0.00 | 5.88 | 0.00 | 0.42 |
| Degraded: thicket and bushland (etc) | 0.11 | 0.63 | 0.00 | 0.00 | 0.61 | 7.50 | 0.02 | 3.12 | **36.65** | 4.77 |
| Degraded: unimproved grassland | 0.00 | 0.00 | 0.00 | 0.00 | 3.73 | 2.82 | **11.02** | 0.49 | 0.00 | 2.93 |
| Degraded: shrubland and low fynbos | 0.76 | 7.66 | 0.00 | 1.05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Cultivated: permanent - commercial irrigated | **11.70** | 5.20 | 0.00 | 1.77 | 0.03 | 0.00 | 0.00 | 1.55 | 0.00 | 0.01 |
| Cultivated: permanent - commercial dryland | 0.32 | 0.05 | 0.00 | 0.00 | 0.00 | 0.39 | 0.01 | 1.78 | 0.00 | 0.00 |
| Cultivated: permanent - commercial sugarcane | 0.00 | 0.00 | 0.00 | 0.00 | **10.79** | **15.39** | 0.00 | 0.34 | 0.00 | **8.91** |
| Cultivated: temporary - commercial irrigated | 0.15 | 2.78 | 0.02 | 2.17 | 1.67 | 0.02 | 0.05 | 2.55 | 0.00 | 0.23 |
| Cultivated: temporary - commercial dryland | **74.78** | **39.53** | **64.65** | **53.07** | 4.74 | 0.00 | **19.58** | 1.30 | 0.00 | 0.49 |
| Cultivated: temporary - semi-commercial / subsistence dryland | 0.00 | 0.00 | 0.00 | 0.00 | 7.02 | **10.18** | **21.27** | **11.80** | **11.40** | **13.75** |
| Forest plantations | 0.60 | 4.88 | 0.00 | 0.31 | **30.86** | **9.31** | 0.06 | **15.29** | 0.00 | **9.11** |
| Urban / built-up land: residential | 1.59 | 7.11 | 0.36 | 0.78 | 0.83 | 3.10 | 0.79 | 1.30 | 0.01 | 1.98 |
| Urban / built-up land: residential (small holdings: woodland) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Urban / built-up land: residential (small holdings: bushland) | 0.00 | 0.00 | 0.00 | 0.00 | 0.14 | 0.90 | 0.00 | 0.00 | 0.00 | 0.04 |
| Urban / built-up land: residential (small holdings: shrubland) | 0.45 | 1.03 | 0.00 | 0.02 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Urban / built-up land: residential (small holdings: grassland) | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.04 | 0.00 | 0.00 | 0.15 |
| Urban / built-up land: commercial | 0.06 | 0.20 | 0.00 | 0.00 | 0.02 | 0.13 | 0.00 | 0.02 | 0.00 | 0.06 |
| Urban / built-up land: industrial / transport | 0.03 | 0.56 | 0.02 | 0.07 | 0.00 | 0.33 | 0.01 | 0.01 | 0.00 | 0.15 |
| Mines & quarries | 0.07 | 0.00 | 0.01 | 0.00 | 0.00 | 0.06 | 0.00 | 0.03 | 0.01 | 0.02 |

*Bold values indicate main land uses in the vegetation type*

The vegetation types listed at the bottom of Table 4 are less impacted on by land uses, and are generally better protected (Table 3), with the Mopane Shrubveld and Thorny Kalahari Dune Bushveld including 100 and 99.6% protected area, respectively. These areas also contain extensive tracts of natural vegetation ranging from 83.5% for the Thorny Kalahari Dune Bushveld to 100% for the Mopane Shrubveld (Table 3). This however does not preclude them from further analysis and the tools developed in this study have a potential role to play in the monitoring and future management of these currently less impacted areas.

*Comparison of vulnerability status*

Low and Rebelo (1996) also provided an estimate of threat status of the vegetation types. This included a measure of land transformed by agriculture and other uses, based on "scant information for some of the Acocks Veld Types and should be cautiously interpreted as a rough index of habitat loss" (Low & Rebelo, 1996). They also include an estimate of the proportion of each vegetation type falling within conserved areas, based on an approximation of conservation area boundaries which still require confirmation (Low & Rebelo, 1996). Following a similar methodology to Thompson *et al.* (in review), we evaluate these estimates from Low and Rebelo (1996) as well as the calculations of protected and transformed land obtained from this study using the National Land-cover database and the DEAT (1996) protected area database (Table 3). Top conservation priority vegetation types identified based on Low and Rebelo's (1996) estimates of transformed area in Table 3 highlight the West Coast Renosterveld, Short Mistbelt Grassland, Coast-Hinterland Bushveld, Natal Central Bushveld and the Moist Clay Highveld Grassland as areas of conservation concern due to large areas transformed. The Mopane Shrubveld, Grassy Fynbos, Mopane Bushveld, Central Mountain Renosterveld and Mountain Fynbos are estimated to be areas of low priority for conservation as they are little transformed according to Low and Rebelo's (1996) estimates (Table 3). Once again the areas of high threat are estimated by Low and Rebelo (1996) to be poorly protected with less than 4% of their surface area protected and those that are low priorities are seen to be generally well protected.

As found in Thompson *et al.* (in review), there is some degree of similarity in the rank orders of vegetation types according to threat status found in this study (i.e., affected area) and in Low and Rebelo's (1996) (i.e., areas estimated to be transformed) ($r_s = 0.55$; $p < 0.001$). However, as Table 3 illustrates, there are differences between these estimates of transformation and protection from Low and Rebelo (1996) and values generated in this study. The Low and Rebelo (1996) estimates for land transformation and protection being consistently and significantly higher (paired t-test for levels of transformation, $t = 9.00$, degrees of freedom = 49, $p < 0.0001$; paired t-test for levels of protection, $t = 3.8$, degrees of freedom = 67, $p < 0.01$). This could however be explained by the fact that the estimates of transformation in Low and Rebelo (1996) included grazed areas, while the NLC transformation category does not (Thompson *et al.* in review). The grazed areas (especially overgrazed area) are included in the degraded category of the NLC database and as such are included in the present study in the measure of affected areas (Table 4).

**Conclusion**

South Africa, with its large biodiversity conservation responsibility, faces the additional problems of limited resources for conservation as well as pressing land reform initiatives. The land tenure system is a problem for conservation throughout Africa and is now becoming an increasingly demanding problem in South Africa. The almost total transfer of land in most regions of South Africa, from government to private ownership, is possibly unique in the annals of European colonisation. The state by the mid 1930's had lost control over resources which in countries such as Australia or the USA were retained by the authorities because of their unsuitability for agriculture (Christopher, 1982). In effect the absence of state interest in land through a leasehold system has lead to a strong demand for land and an attempt to make a living in areas highly unsuitable for the purposes of farming. Demand for land has further driven land prices to levels far in excess of its value as an agricultural commodity.

Therefore the limited resources of available government land and funding need to be efficiently applied in order to ensure effective conservation as well as development opportunities. This investigation provides an important first approximation towards identifying areas where these limited resources should be concentrated by identifying vegetation types with high levels of current and potential anthropogenic land use and inadequate conservation efforts in order to constrain future spreading of transformation. As Rebelo (1997) points out, few vegetation units are spatially uniform in terms of species composition and ecosystem processes, thus further study within these priority areas is required to identify representative conservation sites within these types. Although Low and Rebelo (1996) provided rough estimates of areas considered to be facing high threats, the value of timely land-cover information on the decision making ability for planning is evident from the present study. The advent of the National Land-cover database has provided a much-needed standardised dataset of current land-cover to significantly improve South African land use and conservation planning.

Further issues relevant to the identification of priority conservation areas are the scale of conservation priority setting, and the effects of global climate change on southern African vegetation. Rebelo (1997) points out that generally vegetation types shared with other neighbouring nations are more adequately conserved than vegetation endemic to South Africa. Thus a classification of vegetation types across political boundaries, as well as international co-operation are urgent requirements for future priority setting. In addition to this, future conservation strategies will have to consider the effects of climate change on biodiversity (Rutherford *et al.*, 2000). Not much is known on what these climate changes or their biological impacts will be, but recent work has highlighted a general eastward shift in South African species distributions as areas in South Africa dry out and warm up (Rutherford *et al.*, 2000; van Jaarsveld & Chown, 2000; van Jaarsveld *et al.*, 2000). It has also been shown that premier flagship conservation areas in South Africa are not likely to meet their conservation goals due to an inability to track climate induced species (especially vulnerable species) range shifts (van Jaarsveld *et al.*, 2000). This is of obvious importance in any conservation-planning scenario.

In many respects "lines conquer", and the South African landscape is a testament to their power. Compasses and plumblines, more than a force of arms, subdue landscapes, and henceforth demarcate control and change. If current development policies (i.e., Spatial Development Initiatives, unstructured land reform) continue without proper equity towards conserving the most threatened vegetation communities, in a few decades not only will the remaining "natural" areas be gone, but the people will be even poorer for it.

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## References

Acocks, P.H. (1988). Veldtypes of South Africa, 3rd Edition. *Memoirs of the Botanical Survey of South Africa*. **57**, 1-146.

Bedward, M., Pressey, R.L. & Keith, D.A. (1992). A new approach for selecting fully representative reserve networks: addressing efficiency, reserve design and land suitability with an iterative analysis. *Biological Conservation*. **62,** 115-125.

Central Statistical Survey. (1998) The People of South Africa Population Census, 1996. Report No. 1:03-01-11, Pretoria.

Christopher, A.J. (1982) South Africa. Longman Press, London.

Conservation International. (1998) Conservation International lists the countries of megadiversity in the world. Http://www.conservation.org.

DEAT. (1996) *National Protected Areas Database*. Department Environmental Affairs and Tourism, Pretoria.

Fairbanks, D.H.K. & Benn, G.A. (2000) Identifying regional landscapes for conservation planning: A case study from KwaZulu-Natal, South Africa. *Landscape and Urban Planning*. **50**, 1-19.

Fairbanks, D.H.K. & Thompson, M.W. (1996) Assessing land-cover map accuracy for the South African land-cover database. *South African Journal of Science*. **92**, 465-470.

Fairbanks, D.H.K., Thompson, M.W.M., Vink, D.E., Newby, T. S., van den Berg, H.M. & Everard, D.A. (2000) The South African land-cover characteristics database: a synopsis of the landscape. *South African Journal of Science*. **96**, 69-82.

Forman, R.T.T. (1998) Road ecology: A solution for the giant embracing us. *Landscape Ecology* **13,** iii-v.

Forman, R.T.T. (2000) Estimate of the area affected ecologically by the road system in the United States. *Conservation Biology*. **14,** 31-35.

Forman, R.T.T. & Alexandra, L.E. (1998) Roads and their major ecological effects. *Annual Reviews of Ecology and Systematics*. **29**, 207-231.

Franklin, J.F. & Forman, R.T.T. (1987) Creating landscape patterns by forest cutting: ecological consequences and principles. *Landscape Ecology*. **1,** 5-18.

Freitag, S., Nicholls, A.O. & Van Jaarsveld, A.S. (1996) Nature reserve selection in the Transvaal, South Africa: what data should we be using? *Biodiversity Conservation*. **5**, 685-698.

Hoffmann, M.T. (1997) Human impacts on vegetation. *Vegetation of southern Africa*. (eds. R.M. Cowling, D.M. Richardson and S.M. Pierce), pp 507-534. Cambridge University Press, UK.

Low, A.B. & Rebelo, A.G. (1996*) Vegetation Map of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs, Pretoria, South Africa.

Maddock, A. & Du Plessis, M.A. (1999) Can species data only be appropriately used to conserve biodiversity. *Bioiversity & Conservation*. **8**, 603-615.

Margules, C.R. & Pressey, R.L. (2000) Review article: Systematic conservation planning. *Nature*. **405**, 243-253.

McDonald, D.J. (1997) VEGMAP: a collaborative project for a new vegetation map of southern Africa. *South African Journal of Science*. **93**, 424-426.

McNeely, J.A. (1994) Protected areas for the 21$^{st}$ century: working to provide benefits to society. *Biodiversity and Conservation*. **3**, 390-405.

Milton, S.J. & MacDonald, I.A.W. (1988) Tree deaths near tar roads in the Northern Transvaal. *South African Journal of Science*. **84,** 164-165.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., de Fonesca, G.A.B. & Kent, J. (2000) Biodiversity hotspots for conservation priorities. *Nature*. **403**, 853-58.

Noss, R.F. (1983). A regional landscape approach to maintaining diversity. *BioScience*. **33,** 700-706.

Noss, R.F. (1987) From plant communities to landscapes in conservation inventories: A look at The Nature Conservancy (USA*)*. *Biological Conservation*. **84**, 223-234.

Noss, R.F. (1990) Indicators for monitoring biodiversity: a hierarchial approach. *Conservation Biology*. **4**, 355-64.

Panzer, R. & Schwartz, M.W. (1998). Effectiveness of a vegetation-based approach to insect conservation. *Conservation Biology* **12,** 693-702.

Pressey, R.L. (1994) Land classifications are necessary for conservation planning but what do they tell us about fauna. *Future of the Fauna of Western New South* Wales (Eds. D. Lunney, S. Hand, P. Reed & D. Butcher), pp 31-41. The Royal Zoological Society of New South Wales, Mosman, Australia.

Pressey, R.L., Humphries, C.J., Margules C.R., Vane-Wright R.I. & Williams P.H. (1993) Beyond opportunism: key principles for systematic reserve selection. *TREE*. **8**, 124-28.

Pressey, R.L. & Logan, V. S. (1994) Level of geographic subdivision and its effects on assessments of reserve coverage: a review of regional studies. *Conservation Biology*. **8**, 1037-46.

Pringle, J. (1982) *The Conservationists and the Killers*. T.V. Bulpin and Books of Africa, Cape Town.

Rebelo, A.G. (1997) *Vegetation of southern Africa*. (eds. R.M. Cowling, D.M. Richardson and S.M. Pierce), pp 571-590. Cambridge University Press, UK.

Reijen, R., Foppen, R., Ter Braak, C. & Thissen, J. (1995) The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to the proximity of main roads. *Journal of Applied Ecology*. **32,** 187-202

Reyers, B., van Jaarsveld, A.S., & Kruger, M. (2000) Complementarity as a biodiversity indicator strategy. *Proceedings of the Royal Society of London: B*. **267**, 505-13.

Rutherford, M.C., Powrie, L.W. & Schulze, R.E. (2000) Climate change in conservation areas of South Africa and its potential impact on floristic composition: a first assessment. *Diversity and Distributions*. **5,** 253-262.

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, H., Caicco, S., D'erchia, F., Edwards, T.C., Ulliman, J. & Wright, R.G. (1993) Gap analysis: A geographic approach to protection of biological diversity. *The Journal of Wildlife Management: Wildlife Monographs No. 123.* **57**, 1-41.

Siegfried, W.R., Benn, G.A., & Gelderblom, C.M. (1998) Regional assessment and conservation implications of landscape characteristics of African national parks. *Biological Conservation.* **84**, 131-140.

Soulé, M.E. & Sanjayan, M.A. (1998). Conservation targets: do they help? *Science* **279**, 2060-2061.

Stoms, D.M. (2000) GAP management status and regional indicators of threats to biodiversity. *Landscape Ecology.* **15**, 21-33.

Thompson, M. (1996) The standard land-cover classification scheme for remote-sensing application in South Africa. *South African Journal of Science.* **92**, 34-42.

Thompson, M., Vink, E.R., Fairbanks, D.H.K., Balance, A. & Shackelton, C.M. In Review. Comparison of Extent and Transformation of South Africa's Woodland Biome from Two National Databases. *South African Journal of Science.*

Trombulak, S.C. & Frissell, C.A. (2000) Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology.* **14**, 18-30.

Van Jaarsveld, A.S. (2000) Biodiversity: conserving the building blocks of environmental health. *The sustainability challenge for southern Africa.* (ed. J. Whitman), pp 265-290. Macmillan Press Ltd, London, UK.

Van Jaarsveld, A.S. & Chown, S.L. (2000). Climate change and its impacts in South Africa. *TREE.* **16**, 13-14.

Van Jaarsveld, A.S., Chown, S.L., Erasmus, B.F.N., Kshatriya, M. & Wessels, K.J. (2000) *Vulnerability and adaptation assessment of South African animal taxa to climate change.* Report to the Department of Environmental Affairs and Tourism, Pretoria, South Africa

Van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998) Biodiversity assessments and conservation strategies. *Science.* **279**, 2106-2108.

Wessels, K.J., Freitag, S. & van Jaarsveld, A.S. (1999) The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological Conservation.* **89**, 21-38.

WCMC. (1992) *Development of a National Biodiversity Index: a discussion paper prepared by World Conservation Monitoring Centre. Report on the WCMC*, 15 September, 1992.

Williams, P.H. (1998) Key sites for conservation: area selection methods for biodiversity. *Conservation in a Changing World: Integrating Processes into Priorities for Action.* (Eds. G.M. Mace, A. Balmford & J.R. Ginsberg) pp 211-249. Cambridge University Press, UK.

Williams, P.H. & Humphries, C.J. (1996) Comparing character diversity among biotas. *Biodiversity: a biology of numbers and difference.* (Ed. K.J. Gaston). Pp 54-76. Blackwell Science, Oxford.

WRI. (1994) *World Resources 1994-1995.* Oxford University Press, Oxford, UK.

# ADDENDUM II

## Incorporating land-cover information into regional biodiversity assessments in South Africa.

# Incorporating land-cover information into regional biodiversity assessments in South Africa.

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**Abstract**

Anthropogenic natural habitat transformation presents the single most important threat to global biodiversity. Land-cover data, based on Landsat TM imagery, were used to derive land-use information for the Gauteng, Mpumalanga and Northern provinces of South Africa. The assessment integrated land-use data with species presence data (15' x 15' grid cell resolution) for butterflies, mammals, birds and endemic vascular plants. The objectives of the present study were: (i) to identify areas at a regional scale where there is a possible conflict between biodiversity conservation interests and current land-uses; (ii) to investigate the influence of incorporating a land-use constraint (LUC) into a conservation area selection algorithm, while taking cognizance of the existing reserve system; and (iii) to investigate the circumstances of species recorded within these conflict areas. Many grid cells identified as species richness hotspots, rarity hotspots or as part of the complementarity network selected by the unconstrained algorithm were in reality largely transformed or modified. These areas should thus be avoided when striving to identify a viable conservation network. Although the LUC algorithm selected more grid cells to represent all species, it succeeded in increasing the percentage natural vegetation within the selected conservation network and highlighted areas where potential conflicts should be thoroughly investigated at a local scale.

## Introduction

Land-cover refers to the suite of natural and man-made features that cover the earth's immediate surface (Thompson, 1996). Natural land-cover represents the green interface between the lithosphere and the atmosphere that has a profound influence on the climate and biogeochemical systems and forms the basic fabric of biodiversity (Graetz, Fisher & Wilson, 1992). Land-cover changes, caused by increases in crop cultivation and urban development, present the single most important threat to global biodiversity (Soulé, 1991; Dale *et al.*, 1994). Habitat destruction, as a direct consequence of human activity, accounts for the fact that current species extinction rates exceed historical global extinction rates by between 1000 and 10000 times (Wilson, 1988; UNEP, 1995). Macdonald (1989) estimated that up to 25% of South Africa's natural land-cover has been converted to other forms of land-use such as agriculture, which accounts for more than half of that transformation.

As signatories to the Convention of Biodiversity, South Africa is obligated to: "Review the impact of agriculture and commercial forestry practices on biodiversity (natural habitats) and seek changes where necessary" (DEAT, 1996). Satellite remote sensing has provided us with an effective tool for gathering this essential land-cover information (Dale *et al.*, 1994; Scott *et al.*, 1993). Land-cover data, generated by the Agricultural Research Council (ARC - Institute for Soil, Climate and Water) and the CSIR (Council for Scientific and Industrial Research), recently became available for South Africa. Although land-cover and land-use are not necessarily synonymous (Thompson, 1996) broad land-use categories (e.g. cultivation, urban or natural vegetation) can be derived from satellite derived land-cover data.

Existing protected areas were primarily proclaimed on an *ad hoc* basis and are mostly ineffective at representing regional biota's (Pressey, 1994; Lombard, 1995a,b). In response, systematic reserve selection procedures were developed to identify priority conservation areas that complement one another in terms of their contributions towards protecting regional biodiversity, while ensuring that minimal land allocation is required (Margules, *et al.*, 1988; Nicholls & Margules, 1993; Pressey *et al.*, 1993; Margules, Cresswell & Nicholls, 1994; Csuti *et al.*, 1997; Lombard, 1995a; Freitag, Nicholls & van Jaarsveld; 1996, Wessels, Freitag & van Jaarsveld, 1999). Within South Africa several national and regional biodiversity assessments based on historical species presence data of specific taxa within 15' x 15' grid cells have been conducted, including fish (Skelton *et al.*, 1995); frogs (Drinkrow & Cherry, 1995); tortoises (Branch, Benn & Lombard, 1995); snakes (Lombard, Nicholls & August, 1995); mammals (Gelderblom *et al.*, 1995; Mugo *et al.*, 1995; Gelderblom & Bronner, 1995; Freitag *et al.*, 1996); birds (Lombard, 1995a); plants (Rebelo & Siegfried, 1992); and multiple taxa including birds, mammals, insects and plants (van Jaarsveld *et al.*, 1998a). It is however possible that an area (e.g. grid cell) selected for its contribution to species representation, according to historical data, may in reality be largely transformed by extant land-uses. For this reason a number of previous studies have used aerial photographs (Awimbo, Norton & Overmars, 1996; Lombard *et al.*, 1997), NOAA (Bull, Thackway &

Cresswell, 1993) and Landsat TM satellite images (Bedward, Pressey & Keith, 1992; Scott *et al.*, 1993; Pressey *et al.*, 1996) to map transformed areas and exclude these during conservation area selection.

Although specific species may persist within the altered landscape mosaic of a highly transformed grid cell (Soulé, 1991; Jules & Dietsch, 1997; Vandermeer & Perfecto, 1997), the long-term survival of all native species is ultimately determined by a complex interaction between (i) the susceptibility of individual species to extirpation, i.e. life-history, gap-crossing ability, area requirements (Dale *et al.*, 1994; White *et al.*, 1997), (ii) local scale landscape pattern, i.e. availability, diversity, fragmentation, spatial configuration, patch size of natural habitat (Lovejoy *et al.*, 1983; Freemark, 1995; Allan *et al.*, 1997; van Jaarsveld, Ferguson & Bredenkamp, 1998b; Brokaw, 1998) , (iii) the nature and environmental impact of interspersed alternative land-uses, i.e. land-use diversity, intensity, and the impact of e.g. agricultural or forestry practices on hydrological processes and soil properties (Hobbs, 1993; McFarlane, George & Farrington, 1993; Nulsen, 1993; Saunders *et al.*, 1993; Freemark, 1995; Smith, 1996; Jules & Dietsch, 1997) and (iv) degradation within natural areas, e.g. overgrazing of rangelands (Grant *et al.*, 1982; Barnes, 1990; O'Connor, 1991; Scholtz & Chown, 1993; Srivastava, Smith & Forno, 1996a, Joubert, 1998; Seymour, 1998; Todd & Hoffman, In Press). Although highly transformed areas may currently harbor certain species, these may not sustain natural ecological processes and complete samples of other non-target taxa (Baudry, 1993; Di Benedetto *et al.*, 1993; Hobbs, 1993; Freemark, 1995), thus largely precluding these areas from feasible regional conservation networks.

The objectives of the present study were: (i) to identify areas at a regional scale where there is a possible conflict between biodiversity conservation interests and current land-uses; (ii) to investigate the influence of avoiding such potential conflict areas by incorporating a land-use constraint (LUC) into a conservation area selection algorithm, while simultaneously taking cognizance of the existing reserve system; and (iii) to investigate the circumstances of species recorded within these conflict areas.

## Methods

### *Study area*

The study area comprised the Gauteng, Mpumalanga and Northern provinces of South Africa (Figure 1) and represents 17.3% (219180 $km^2$) of the land area of one of the most biologically rich countries in the world (WCMC, 1992). The study area includes three of South Africa's seven biomes, namely grasslands, savanna and forests (Low & Rebelo, 1996).

### *Species distribution data*

Information on historically recorded species presence within 15' x 15' grid cells (~ 26km x 26km; hereafter referred to as grid cells) was collated for butterflies (Lepidoptera: superfamilies Hesperiodea, Papilionoidea), mammals, birds and endemic vascular plants (van Jaarsveld *et al.*, 1998a). According to

Harrison (1992) the bird data reflect no survey bias. Although the butterfly dataset contains the fewest number of records (Table 1), it represents the best available insect dataset for the study area (Muller, 1999). The mammal database incorporates all terrestrial orders and contains no fundamental sampling bias within the study area (Freitag & van Jaarsveld, 1995; Freitag *et al.*, 1998). Only endemic plant species (i.e. species that have not been recorded outside the study area in South Africa) were included in this analyses, since the representation of all plant species set outrageous conservation demands, i.e. 50% of total area (unpublished). Plant data were available for all grid cells in the study area, but only 87% of them contained records of endemic species (Table 1). These data represent the most comprehensive regional biodiversity data currently available for South Africa (van Jaarsveld *et al.*, 1998c).

*Land-cover data*

Land-cover data were mapped (using manual photo-interpretation) from 1:250000 scale geo-rectified space-maps, based on seasonally standardized LANDSAT Thematic Mapper satellite imagery captured primarily during 1994-95 (Thompson, 1996). For the purpose of the present study, the 31 land-cover classes were reclassified into three categories, namely natural vegetation, modified vegetation and transformed (Table 2). Natural vegetation included all untransformed vegetation, e.g. forest, thicket and grassland. The modified vegetation category was dominated by various "degraded" classes and also included waterbodies (mostly dams) (Thompson, 1996). The degraded classes included all areas with very low vegetation cover in comparison with the surrounding natural vegetation cover and were typically associated with subsistence level farming and rural population centres, where wood-resource removal, overgrazing and subsequent soil erosion were excessive (Thompson, 1996). Transformation was defined as changes to the natural ecosystems in which the structure and species composition were completely or almost completely altered (Poore, 1978). The transformed category therefore encompassed all the cultivated and urban/built-up classes, forestry plantations (mainly commercial *Pinus* and *Eucalyptus* species), as well as mines and quarries (Macdonald, 1989).

*GIS analysis*

The land-cover data for the study area were overlaid with a 15' x 15' grid. Only grid cells that overlapped at least 20% with the study area were included in the analyses (n = 336; Figure 1). The extent of the protected areas (provincial and national parks) and various land-cover classes within each grid cell were calculated using ArcInfo (Albers equal area projection).

*Conservation Area Selection*

Richness and rarity hotspots were identified within the study area. Richness hotspots were defined as the top five percent (n = 17) species-rich grid cells, whereas rarity hotspots were all grid cells containing database rare species (< 1% of grid cells; n ≤ 3) (van Jaarsveld *et al.*, 1998a).

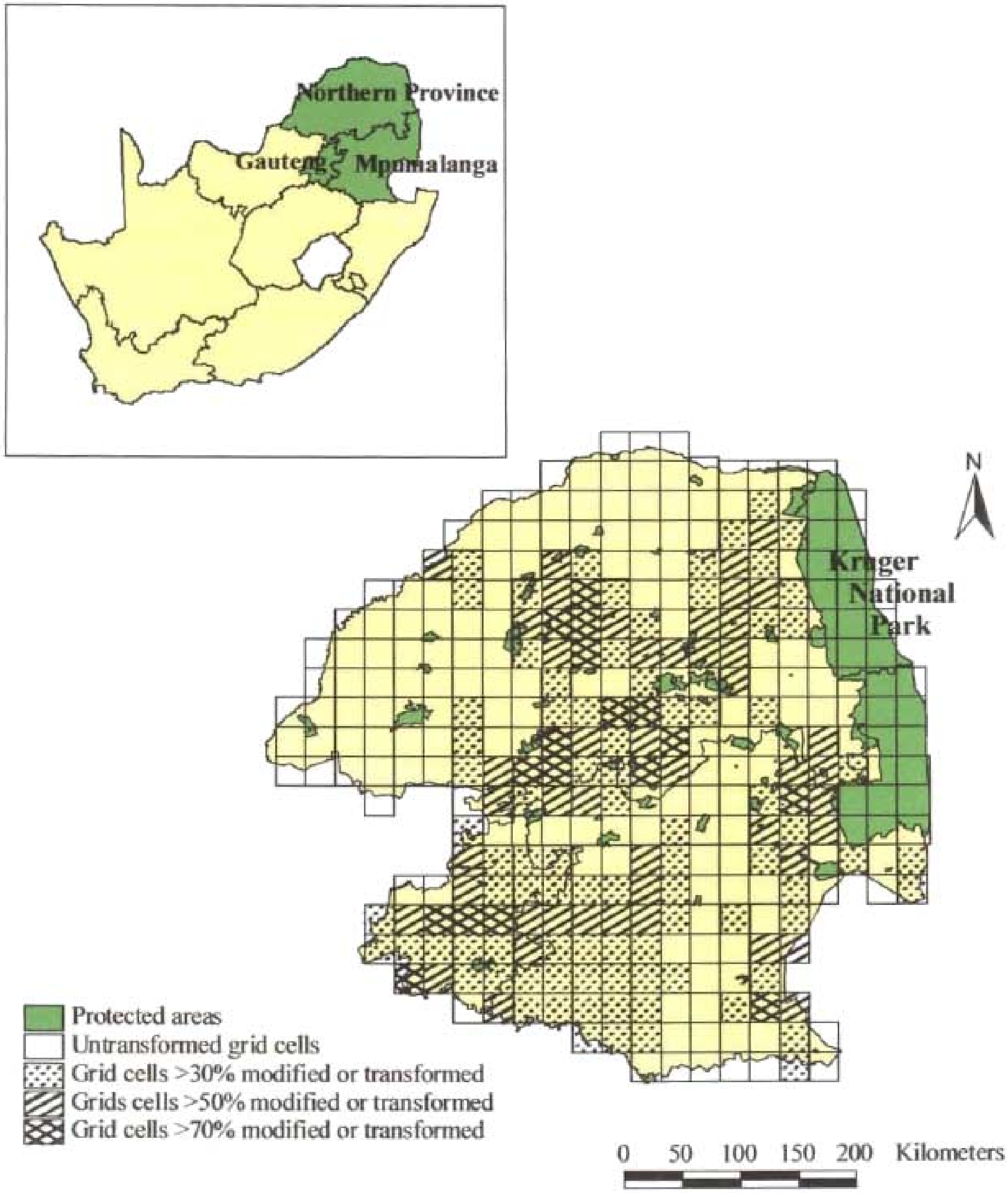


**Figure 1:** Three provinces of South Africa (see inset), with protected areas (Provincial and National parks), overlaid with a 15' x 15' grid and showing cells that are more than 30, 50 and 70% modified or transformed.

**Table 1:** Information on species presence data.

| Taxa | Number of records | Time span | Number of species | Number of grid cells with records | Rare species, in less than 1% of grid cells |
|---|---|---|---|---|---|
| Birds | 79082 | 1980-95 | 581 | 336 (100%) | 25 (4.3%) |
| Butterflies | 3 725 | 1900-80 | 369 | 142 (42%) | 92 (24%) |
| Endemic plants | 4451 | 1900-96 | 366 | 295 (87%) | 112 (30.6%) |
| Mammals | 5929 | majority after 1980 | 191 | 268 (79%) | 32 (16.8%) |
| **Total** | **93187** | | **1507** | **336 (100%)** | **261 (17.3%)** |

**Table 2.** Land-cover classes reclassified into categories and the percentage of the study area covered by each category.

| Land-cover category | % area | Land-cover classes |
|---|---|---|
| natural vegetation | 70.7% | forest and woodland; forest; thicket, bushland; shrubland and low fynbos; herbland; grassland; wetlands. |
| modified vegetation | 6.6% | all degraded classes (6.2%);<br>waterbodies (0.3%). |
| transformed | 22.7% | all cultivated classes (15.7%);<br>all urban/built-up classes (2.8%);<br>mines and quarries (0.4%);<br>forest plantations (3.8%). |

This defined rarity could be the consequence of a restricted range or inadequate sampling effects (Gaston, 1991). Complementary sets representing all species at least once, were identified using a rarity-based algorithm that included an adjacency constraint (Nicholls & Margules, 1993). To take the contribution of existing national and provincial parks into account (Figure 1), all species occurring in one or more grid cells that overlap more than 90% with protected areas, were treated as already represented and were excluded from the selection procedures.

To identify a conservation area network that reduces conflict with other land-uses, the algorithm was modified to include constraints that successively exclude from selection grid cells that are more than 10, 20, 30...90% transformed or modified (Lombard *et al.*, 1997). In essence, the land-use constrained (LUC) algorithm was initially limited to select only grid cells that were less than 10% modified or transformed until no new species could be added to the system. After that it proceeded in a step-wise fashion to select grid cells that are more than 10, 20, 30 ...90% modified or transformed, until all species were represented. The LUC algorithm was therefore based on a trade-off between the primary objective of avoiding transformed land and a secondary objective of minimising the number of grid cells required to represent all species, i.e. maximising efficiency (Pressey *et al.*, 1993; Nantel *et al.*, 1998).

**Results**

Table 2 provides the percentages of the study area covered by the three land-cover categories. Approximately 23% of the study area was transformed, whereas 6.6 % was modified, with degradation accounting for the majority (6.2%) of the latter (Table 2). Figure 1 to Figure 3 illustrate the distribution of grid cells that have been modified or transformed to various degrees.

Of the 17 identified richness hotspots, nine (53%), six (35%) and two (12%) were respectively more than 30, 50 and 70% modified or transformed. 17% (261/1507, Table 1) of the species were recorded in less than one percent of the grid cells. These rare species occurred in 149 rarity hotspots of which 60 (40%), 29 (19%) and six (4%) were more than 30, 50 and 70% modified or transformed.

Seventeen of the grid cells overlapping with the Kruger National Park (2 million ha) fall more than 90% within this protected area (Figures 1–3). These 17 grid cells included at least one record for 772 species, thus leaving 735 species (hereafter referred to as remaining species) to be represented elsewhere.

Figure 4a illustrates the cumulative number of remaining species represented within each grid cell selected by the unconstrained and LUC algorithms. To represent all remaining species (n = 735), the unconstrained algorithm selected 77 grid cells (24% of 319), of which 36 (47%), 20 (26%) and four (5%) were respectively more than 30, 50 and 70% modified or transformed (Figure 2). Species were rapidly added during the first quarter of the unconstrained algorithm's curve, after which progress was slower (Figure 4a). The curve of the LUC algorithm periodically accelerated and slowed down to form distinct steps as the algorithm successively selected from sets of grid cells which were increasingly modified or transformed, at 10% increments (Figure 4a).

Figure 4b illustrates the percentage modification and transformation within each individual grid cell selected by the two algorithms. The unconstrained algorithm showed considerable variation throughout the entire curve, with no apparent trend (Figure 4b). The LUC algorithm's curve (Figure 4b) displayed some variation within each of the steps and clearly illustrates its attempt to near-minimise the extent of modified or transformed areas within the grid cells selected.

Figure 5 illustrates the land-use scenarios within the grid cells selected by the unconstrained and LUC algorithms. The complete reserve network selected by the LUC algorithm contained 7.8% more natural habitat than the set selected by the unconstrained algorithm (Figures 5a & 5b). The LUC algorithm required a total of 119 grid cells to represent all remaining species (Figure 3); 54% more than the unconstrained algorithm (n = 77). The LUC algorithm managed to represent 88% of all species within 81 grid cells which were less than 30% modified or transformed (Figures 4a & 4b), with an average of 13% modified or transformed area per grid cell. The LUC algorithm proceeded to represent 95.4% of the species in 102 grid cells which were less than 50% modified or transformed (average of 19% modified or transformed area per grid cell) (Figures 3 & 5c). An additional 17 grid cells, which were more than 50% modified or transformed (average of 60% modified or transformed area per grid cell) were required to represent the deficit of 34 (4.6%) species (Figures 3 & 5d).

## Discussion

### *Land-use scenarios and potential conflict areas*

Since the turn of the century the area of cultivated land in SA has steadily increased from approximately three percent in 1911 to eight percent in 1981 (Scotney *et al.*, 1988). The three provinces (Mpumalanga, Gauteng and Northern province) include extensive areas of arable land and as a result 15.7% of the study area has been transformed by cultivation. Forestry plantations (3.8%) and urban/built-up areas (2.8%) account for the remaining land transformation (Table 2). However, the study area has not been excessively modified or transformed, since 70% is still covered by natural vegetation. Land-uses within areas covered by natural vegetation include wildlife reserves, game ranching and cattle grazing (rangeland), all of which are considered to be amenable to biodiversity conservation (Pressey, 1992). When compared with other biodiversity assessments, where only 34% (Hokitika, New Zealand; Awimbo *et al.*, 1996), 8% (Bega Valley, New South Wales; Keith, 1995) or as little as 7% natural vegetation remains (Western Australian wheatbelt; Saunders, Hobbs & Arnold, 1993), the biodiversity in the present study area does not appear to be in the dire situation prevailing elsewhere.

A number of grid cells identified as species richness hotspots, rarity hotspots or part of the complementary set selected by the unconstrained algorithm (Figure 2), were in reality largely transformed or modified (e.g. around the towns of Sabie, Tzaneen, Graskop, Warmbad; Table 3).

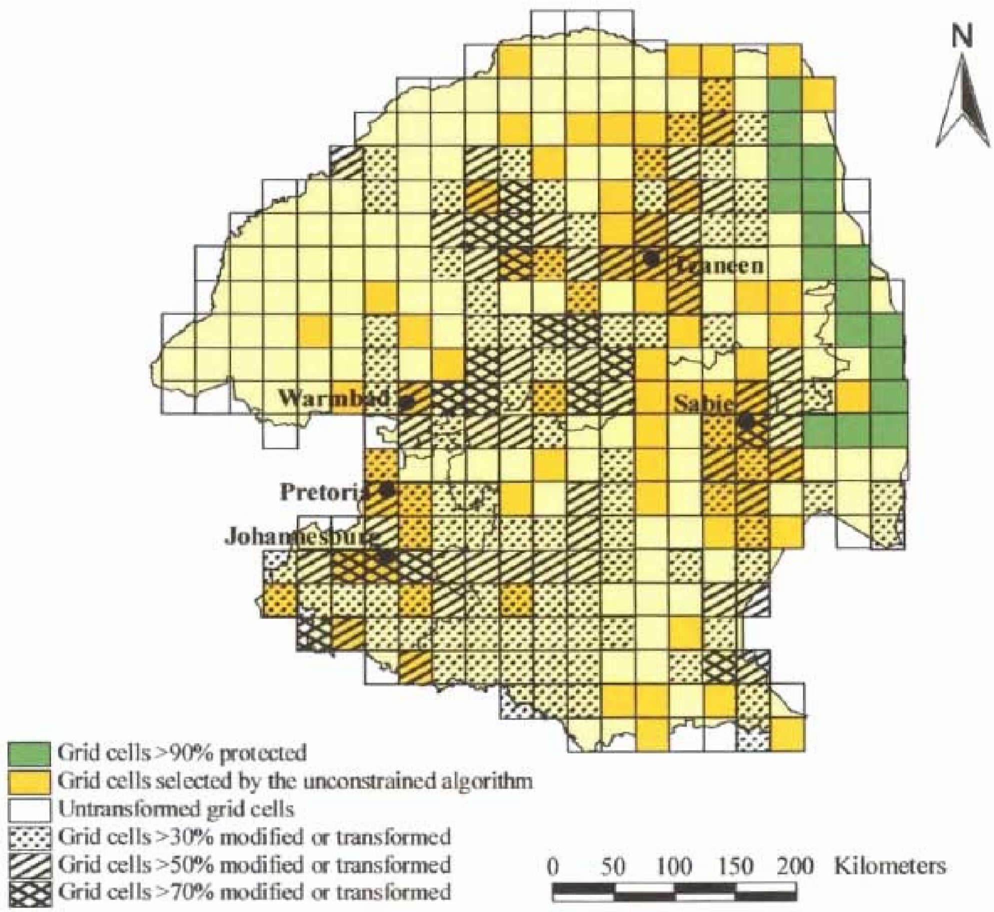


**Figure 2:** Map of study area with grid cells selected by unconstrained complementarity algorithm, while taking cells into account that are more than 90% protected. Overlap with grid cells that are more than 30, 50 and 70% modified or transformed is also included.

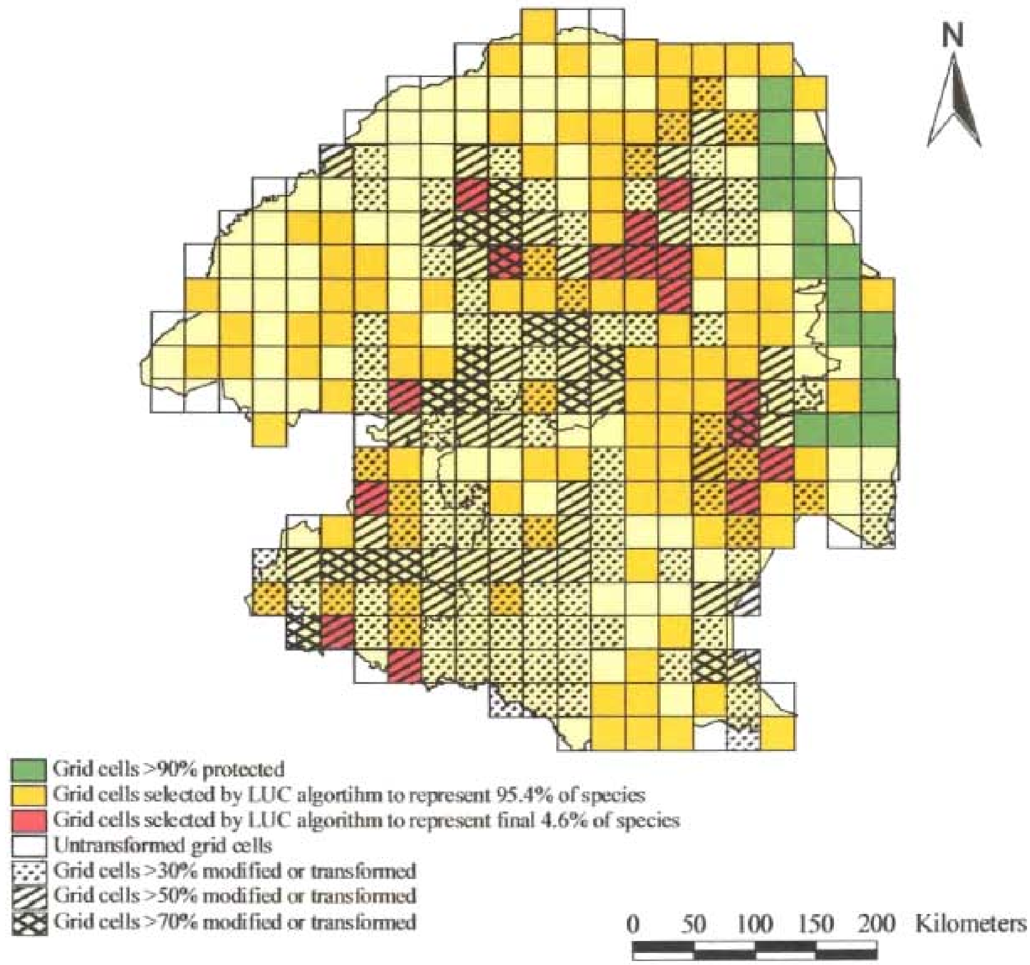


**Figure 3:** Map of study area with grid cells selected by land-use constrained algorithm (LUC), while taking cells into account that are more than 90% protected. The initial 102 cells selected are distinguished from the additional 17 cells that were more than 50% modified or transformed.

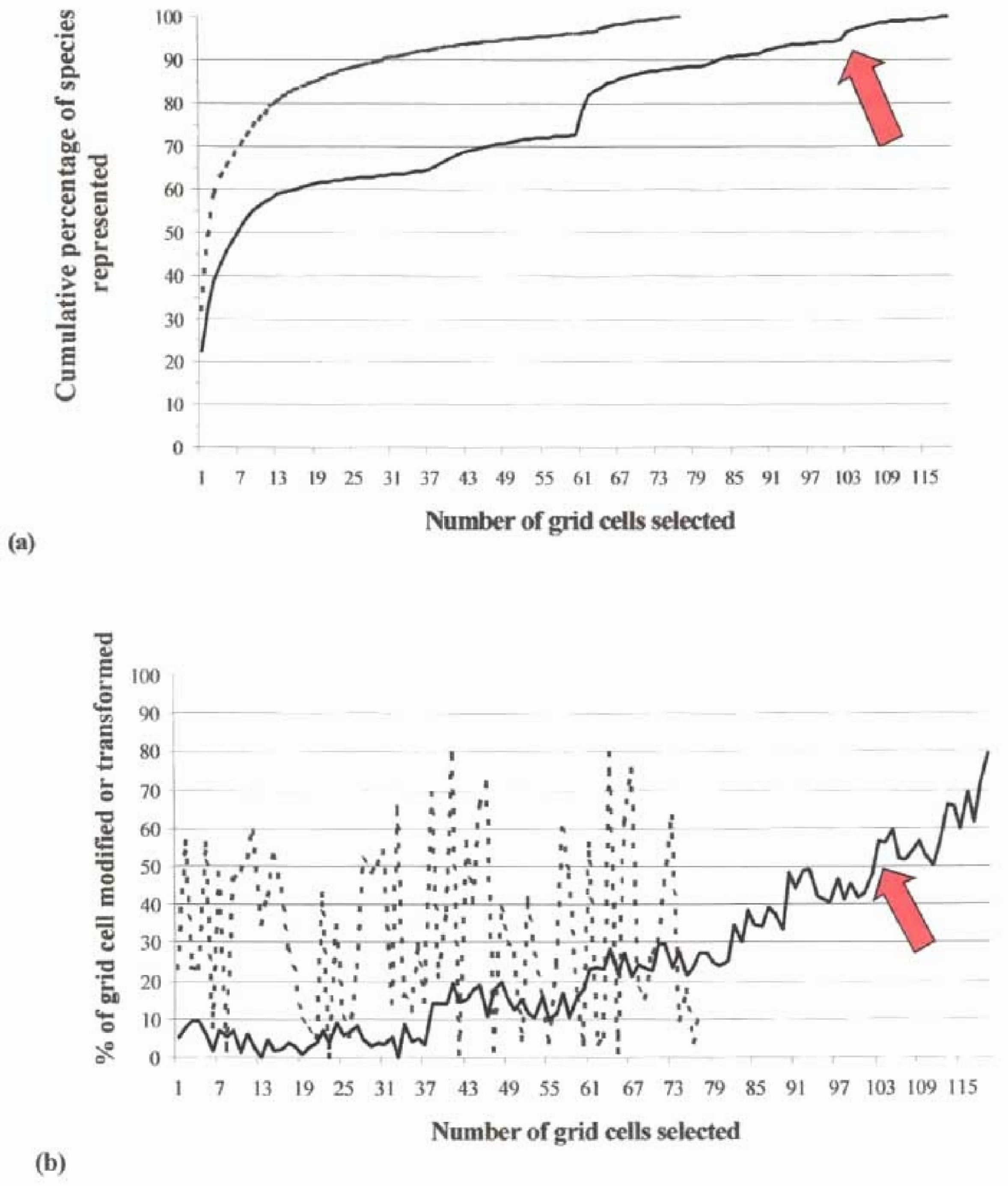


**Figure 4:** Results of the unconstrained (dashed lines) and LUC (solid lines) selection algorithms: (a) Number of species represented within grid cells selected. (b) Percentage area modified or transformed within selected grid cell. Arrows indicate where the LUC algorithm starts selecting grid cells that are more than 50% transformed in order to represent the remaining 4.6% of the species.

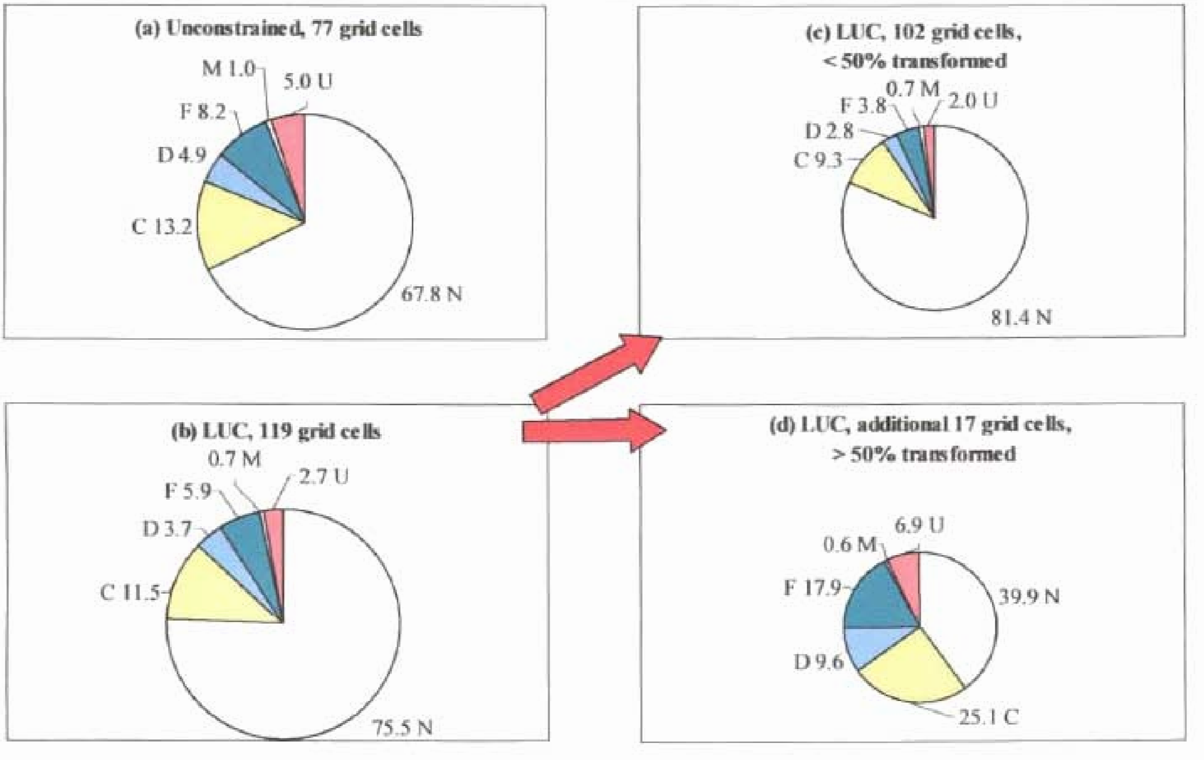


**Figure 5:** Percentage area of specified (a-d) selected sets of grid cells covered by natural vegetation (N), cultivation (C), urban / built-up (U), forestry plantations (F), degradation (D) or other modified land-cover classes (M). (a) Unconstrained algorithm, 77 grid cells representing all species; (b) Land-use constrained (LUC) algorithm, 119 grid cells representing all species; (c) LUC algorithm, 102 grid cells which are less than 50% modified or transformed, representing 95.4% of remaining species; (d) LUC algorithm, 17 additional grid cells which are more than 50% modified or transformed representing 4.6% of remaining species.

**Table 3:** Land-use within a subset of highly transformed grid cells identified as richness hotspots, rarity hotspots or belonging to a complementary set selected by both the unconstrained and the LUC algorithms (Figure 2).

| Grid cell | Cultivated | Forestry | Urban | Degraded and other modified | Natural vegetation |
|---|---|---|---|---|---|
| Sabie | 2% | 76% | 1.5% | 0.5% | 20% |
| Tzaneen | 25.5% | 33% | 2% | 1.5% | 38% |
| Graskop | 1.5% | 57% | 0.5 | 1% | 40% |
| Warmbad | 43% | 0.5% | 3.5% | 5% | 48% |
| Pretoria | 12.5% | 1% | 42.5% | 0.5% | 43.5% |
| Johannesburg | 0.02% | 1% | 67% | 11.08% | 20% |

The most conspicuous of these potential conflict areas are grid cells that coincide with the Johannesburg and Pretoria metropolitan areas (Table 3) (Figure 2). Although these species data ascribe a high conservation value to the above-mentioned transformed areas, these areas may not support natural ecological processes or a complete assemblage of all native species (Baudry, 1993; Di Benedetto *et al.*, 1993; Hobbs, 1993; Freemark, 1995). Therefore, these transformed areas should be avoided when striving to identify an attainable and viable conservation network.

*Comparison of the unconstrained and LUC algorithms*

The seventeen grid cells that were regarded as sufficiently protected (more than 90% overlapping with conservation areas, i.e. Kruger National Park), represented 51% of the 1507 species in the database once. Although the cut-off value of 90% protected is as arbitrary as most other conservation targets, e.g. 10% of all vegetation types (Soulé & Sanjayan, 1998), this stringent precondition is an attempt at maximising the probability that all species recorded within a grid cell are protected.

Figure 4a illustrates the efficiency of the unconstrained rarity-based algorithm (Nicholls & Margules, 1993) at representing the remaining 735 species. However, applying this "naive" algorithm, without taking land-cover data into account, resulted in the selection of grid cells that were highly modified or transformed (Figure 2). Seeking efficiency during reserve selection by minimising land requirements, clearly provided results that were impractical conservation options. In accordance with previous findings (Nantel *et al.*, 1998), the present study illustrated that attempts to avoid conflict with other land-uses entails selecting a larger number of areas (between 40 and 55% more) to achieve the same conservation goals. To increase the percentage natural vegetation within the selected set with 7.8%, required an additional 42 grid cells, thus increasing the percentage of grid cells selected from 24% (77 / 319) to 37% (119 / 319)(Figures 4a, 5a & 5b).

Figures 4a and 4b clearly illustrate how the LUC algorithm compromised efficiency to avoid transformed areas. The LUC algorithm accelerated and slowed down periodically as it attempted to represent the maximum number of species within successive sets of grid cells containing specified areas of modified or transformed land (at 10% increments) (Figure 4a). This is in contrast to the results of the unconstrained algorithm which varied considerably in terms of the extent of modification and transformation in the grid cells selected (Figure 4b).

To represent the final 34 species (4.6%) (Appendix I), the LUC algorithm had no other option but to select 17 highly transformed (more than 50% transformed) grid cells (Figure 5d) which were also selected by the unconstrained algorithm (Figures 2 & 3). The overall effectiveness of the land-use constraint (Figure 5b) at maximising the amount of natural habitat within a selected set of areas depends on the availability of alternative areas for the representation of rarely recorded species. Therefore, this effectiveness would have been higher if the number of rare species recorded within highly transformed areas were lower (Table 1; Appendix I). Whether or not portions of these highly transformed grid cells

(Figure 3) should be included in a protected area network can not be determined from the available coarse scale biodiversity (15' x 15' grid cells) and the land-cover data (1:250000).

These results therefore illustrate how this regional biodiversity assessment can highlight areas where the nature and reality of potential conflict between land-uses and conservation interests should be thoroughly investigated at a local scale (Erhardt, 1985; Herkert, 1991; Delphey & Dinsmore, 1993; Nantel *et al.*, 1998). Although the present study presented a simple method of incorporating land-use (land-cover) information into the conventional reserve selection algorithms (Nicholls & Margules, 1993; van Jaarsveld *et al.*, 1998a) as a constraint, multi-criteria analyses which allow trade-offs between conservation and development, have previously been employed to select protected areas based on the principle of complementarity (Faith & Walker, 1996).

*Species within conflict areas*

The conservation status of species only recorded in grid cells which are more than 50% modified or transformed, are summarised in Appendix I. Many of the butterfly species and one bird species (Burchell's courser, *Cursorius rufus*) are common elsewhere and are therefore not conservation priorities for the study area (Appendix I). It may however, prove useful to include "regional occupancy" and "relative endemism" scores into similar future analyses in order to prioritise species for conservation within a specific study area (Freitag & van Jaarsveld, 1997; Freitag *et al.*, 1998).

Where conflict areas are identified by this regional assessment, crucial habitats within these highly transformed grid cells can be identified and protected to ensure the survival of the specific species. The regional assessment revealed that one of the butterfly species, *Alaena margaritacea* (Wolkberg Zulu), which is listed as vulnerable by the Red Data Book (Henning & Henning, 1989), is currently confined to a single known locality in the Northern province, that is 30% transformed by forestry and 20% degraded. Two other butterfly species, *Coeliades anchises* (One-pip Policeman) and *Deudorix penningtoni* (Pennington's Playboy) which are respectively listed as uncommon and common to the study area (Pringle, Henning & Ball, 1994), have only been recorded in highly transformed or modified areas and therefore warrant further investigation.

The two bat species listed in Appendix I are rare vagrants throughout Africa and are therefore not necessarily conservation priorities within the study area. Within South Africa the Mozambique woodland mouse (*Grammomys cometes*) is restricted to northern KwaZulu-Natal and south-eastern Mpumalanga (Skinner & Smithers, 1990), where more than 47% of the single grid cell in which it has been recorded is transformed by forestry. Of the birds in Appendix I, the stripped flufftail (*Sarothrura affini*) is listed as threatened (Brooke, 1984), while 13 and 42% of its range in the grasslands of the study area has been transformed by agriculture and forestry respectively.

Two of the plant species in Appendix I are listed as rare (Hilton-Taylor, 1996). *Aloe peglerae* (Turk's cap or Mountain Aloe) is listed as rare and only occurs in areas around Pretoria and west of

Johannesburg, which have been in highly transformed by cultivation and urban development. Although *Borassus aethiopum* (Borassus palm) is found elsewhere in Africa, it has a protected status in South Africa (Palgrave, 1983), where isolated plants occur in the intensively cultivated (30%) and degraded (21%) area south of Tzaneen.

This regional biodiversity assessment also allows us to investigate the land-use circumstances within the ranges of other important species. Of the grid cells where the globally threatened blue swallow (*Hirundo atrocaerulea*) has been recorded, only 51% of the original grassland remains, while some 38% is transformed by forestry and 5% by cultivation. Within the study area, the area of occupancy of the globally threatened Southern bald ibis (*Geronticus calvus*) (Collar, Crosby & Statterfield, 1994; Harrison *et al.*, 1997) has been degraded (12%) and also transformed by both cultivation (17%) and forestry (11%). The endangered Juliana's golden mole (*Amblysomus julianae*) is endemic to the study area and has a very limited and fragmented distribution (Skinner, In Press). The type locality of this species has however been almost completely transformed by urban development and sand mining along the eastern outskirts of Pretoria (Bronner, 1995).

Vandermeer & Perfecto (1997) suggested that conservation biologists should start thinking of agroecosystems as legitimate objects of study and begin asking the same questions about agroecosystems they ask of "pristine" or "natural" systems, in an endeavor to preserve biodiversity through sustainable agriculture (Srivastava, Smith & Forno, 1996b; Smith, 1996). Therefore there is an urgent need in South Africa for studies on the effects of various land-uses on biodiversity across a hierarchy of spatial and temporal scales.

**Conclusions**

The benefit of maximizing the area of natural habitat within a selected set of areas by incorporating a land-use constraint, carries the cost of selecting a larger total number of areas (grid cells), while the representation of all recorded species may require some level of protection for crucial habitats within highly transformed areas. It is however, unlikely that all the areas identified in these analyses (Figures 2 & 3) can be formally protected and therefore the long-term conservation of biodiversity also depends on maintaining hospitable environments within managed landscapes (Noss & Harris, 1986; Western, 1989; Soulé, 1991; Pimentel *et al.*, 1992; Pressey & Logan, 1997; White *et al.*, 1997). The regional assessment presented here is an effective tool for identifying areas where the future of specific species may rely on well coordinated "off-reserve" management (Keith, 1995; Pressey *et al.*, 1996).

Moreover, methods are needed for predicting potential impacts of various land-uses on biodiversity across a hierarchy of spatial and temporal scales to make land-use planning both clearer and better informed (Freemark, 1995; White *et al.*, 1997). As rudimentary reserve selection algorithms, based purely on biogeography, evolve into more practical tools by, for example, including land-cover data (Pimm & Lawton, 1998), they should be incorporated into regional land-use planning decision support systems (Ive & Cocks, 1988; Bedward *et al.*, 1992; Pressey *et al.*, 1995), where they could systematically stake a claim for biodiversity.

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**References**

Allan, D.G., Harrison, J.A., Navarro, R.A., van Wilgen, B.W. & Thompson, M.W. (1997). The impact of commercial afforestation on bird populations in Mpumalanga Province, South Africa -insights from bird-atlas data. *Biological Conservation* 79, 173-185.

Awimbo, J.A., Norton, D.A. & Overmars, F.B. (1996). An evaluation of representativeness for nature conservation, Hokitika ecological district, New Zealand. *Biological Conservation* 75, 177-186.

Barnes, D.L. (1990). Survey of grazed and ungrazed grassland in the south-eastern Transvaal Highveld: Potential floristic composition and patterns of degradation. *Journal of the Grasslands Society of South Africa* 7, 223-231.

Baudry, J. (1993). Landscape dynamics and farming systems: Problems of relating patterns and predicting ecological changes. In *Landscape Ecology and Agroecosystems*. (Eds. R.G.H. Bunce, L. Ryszkowski. & M.G. Paoletti), pp 21-40. Lewis Publishers, London.

Bedward, M., Pressey, R.L. & Keith, D.A. (1992). A new approach for selecting fully representative reserve networks: addressing efficiency, reserve design and land suitability with an iterative analysis. *Biological Conservation* 62, 115-125.

Branch, W.R., Benn, G.A. & Lombard, A.T. (1995). The tortoises (Testudinea) and terrapins (Pelomedusidae) of southern Africa: their diversity, distribution and conservation. *South African Journal of Zoology* 30, 91-103.

Brokaw, N. (1998). Fragments past present and future. *Trends in Ecology & Evolution* 13, 382-383.

Bronner, G.N. (1995). *Systematic revision of the golden mole genera Amblysomus, Chlorotalpa and Calcochloris (Inectivora: Chrysochloromorpha; Chrysochloridae)*. Unpublished PhD thesis. University of Natal, Durban, South Africa.

Brooke, R.K. (1984). *South African Red Data Book - birds. South African National Scientific Programmes Report no. 97.* Foundation for Research Development, Pretoria.

Bull, A.L., Thackway, R. & Cresswell, I.D. (1993). Assessing conservation of the major Murray-Darling basin ecosystems. Environmental Resources Information Network (ERIN), Australian Nature Conservation Agency, Canberra.

Collar, N.J., Crosby, M.J. & Statterfield, A.J. (1994). *Birds to watch 2.* Birdlife International, Cambridge.

Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L., Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Huso, M. & Sahr, K. (1997). A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80, 83-97.

Dale, V.H., Pearson, S.M., Offerman, H.L. & O'Neill, R.V. (1994). Relating patterns of land-use change to faunal biodiversity in central Amazon. *Conservation Biology* 8, 1027-1036.

DEAT (1997). *White paper on the conservation and sustainable use of South Africa's Biological Diversity*. Department of Environmental Affairs and Tourism, Pretoria.

Delphey, P.J. & Dinsmore, J.J. (1993). Breeding bird communities of recently restored and natural prairie potholes. *Wetlands* 13, 200-206.

Di Benedetto, L., Luciani, F., Maugeri, G., Poli Marchese, E. & Razzara, S. (1993). Role of natural vegetation in the agricultural Landscape for biological conservation in Sicily. In *Landscape Ecology and Agroecosystems*. (Eds. R.G.H. Bunce, L. Ryszkowski. & M.G. Paoletti), pp 131-138. Lewis Publishers, London.

Drinkrow, D.R. & Cherry, M.I. (1995). Anuran distribution, diversity and conservation in South Africa, Lesotho and Swaziland. *South African Journal of Zoology* 30, 82-91.

Faith, D.P. & Walker, P.A. (1996). Integrating conservation and development: effective trade-offs between biodiversity and cost in the selection of protected areas. *Biodiversity & Conservation* 5, 431-446.

Erhardt, A. (1985). Diurnal lepidoptera: sensitive indicators of cultivated and abandoned grassland. *Journal of Applied Ecology* 22, 849-861.

Freemark, K. (1995). Assessing effects of agriculture on terrestrial wildlife: developing a hierarchical approach for the US EPA. *Landscape & Urban Planning* 31, 99-115.

Freitag, S. & van Jaarsveld, A.S. (1995). Towards conserving regional mammalian species diversity: a case study and data critique. *South African Journal of Zoology* 30, 136-144.

Freitag, S. & van Jaarsveld, A.S. (1997). Relative occupancy, endemism, taxonomic distinctiveness and vulnerability: prioritizing regional conservation actions. *Biodiversity & Conservation* 6, 211-232.

Freitag, S., Hobson, C., Biggs, H.C. & van Jaarsveld, A.S. (1998). Testing for potential survey bias: the effect of roads, urban areas and nature reserves on a Southern African mammal data set. *Animal Conservation* 1, 119-127.

Freitag, S., Nicholls, A.O. & van Jaarsveld, A.S. (1996). Nature reserve selection in the Transvaal, South Africa: what data should we be using? *Biodiversity & Conservation* 5, 685-698.

Gaston, K.J (1991). How large is a species geographic range? *Oikos* 61, 434.

Gelderblom, C.M. & Bronner, G.N. (1995). Patterns of distribution and current protection status of the endemic mammals in South Africa. *South African Journal of Zoology* 30, 127-136.

Gelderblom, C.M., Bronner, G.N., Lombard, A.T. & Taylor, P.J. (1995). Patterns of distribution and current protection status of the Carnivora, Chiroptera and Insectivora in South Africa. *South African Journal of Zoology* 30, 103-115.

Graetz, D., Fisher, R. & Wilson, M. (1992). *Looking back: the changing face of the Australian Continent, 1972-1992*. CSIRO, Australia.

Grant, W.E., Birney, E.C., French, N.R. & Swift, D.M. (1982). Structure and productivity of grassland small mammal communities related to grazing-induced changes in vegetation cover. *Journal of Mammalogy* 63, 248-260.

Harrison, J.A. (1992). The South African Bird Atlas Project: Five years of growth. *South African Journal of Science* 88, 410-413.

Harrison, J.A., Allan, D.G., Underhill, L.G., Heremans, M., Tree, A.J., Parker, V. & Brown, C.J. (1997). *The Atlas of Southern African Birds. Vols 1 and 2*. Birdlife South Africa, Johannesburg.

Henning, S.F. & Henning, G. (1989). *South African Red Data Book - Butterflies. South African National Scientific Programmes Report no. 158*. Foundation for Research Development, Pretoria.

Herkert, J.R. (1991). Prairie birds of Illinois: Population response to two centuries of habitat change. *Ilinois Natural History Survey Bulletin*. 34, 393-399.

Hilton-Taylor, C. (1996). *Red Data List of Southern African Plants. - Strelitzia 4*. National Botanical Institute, Pretoria.

Hobbs, R. (1993). Effects of landscape fragmentation on ecosystem processes in the Western Australian wheatbelt. *Biological Conservation* 64, 193-201.

Ive, J. & Cocks, K.D. (1988). LUPIS: A decision-support system for land planners and managers. In *Desktop planning: microcomputer applications for infrastructure and services planning and management*. (Eds. P.W. Newton, M.A.P. Taylor & R. Sharpe), pp 129-139. Hargreen, Melborne.

Joubert, D.F. (1998). *Small mammal and bird community structure in commercial and communal rangelands in a semi-arid shrubland in Namaqualand, South Africa*. Unpublished MSc dissertation: University of Cape Town, Cape Town, South Africa.

Jules, E.S. & Dietsch, T.V. (1997). Dangers in dividing conservation biology and Agroecology. *Conservation Biology* 11, 1272-1273.

Keith, D. (1995). Involving ecologists and local communities in survey, planning and action for conservation in a rural landscape: an example from the Bega Valley, New South Wales. In *Nature Conservation 4 - the Role of Networks*. (Eds. D.A. Saunders, J.L. Craig. & E.M. Mattiske), pp 385-400. Surrey Beatty, Sydney.

Lombard, A.T. (1995a). The problems with multi-species conservation: do hotspots, ideal reserves and existing reserves coincide? *South African Journal of Zoology* 30, 145-163.

Lombard, A.T. (1995b). Introduction to an evaluation of the protection status of South Africa's vertebrates. *South African Journal of Zoology* 30, 71-82.

Lombard, A.T., Cowling, R.M., Pressey, R.L. & Mustart, P.J. (1997). Reserve selection in a species-rich and fragmented landscape on the Augulhas plain, South Africa. *Conservation Biology* 11, 1101-1116.

Lombard, A.T., Nicholls, A.O., & August, P.V. (1995). Where should nature reserves be located in South Africa? A snake's perspective. *Conservation Biology* 9, 363-372.

Lovejoy, T.E., Bierregreerd, R.O., Rankin, J.M., & Schubart, H.O.R. (1983). Ecological dynamics of tropical forest fragments. In *Tropical Rainforests: Ecology and Management*. (Eds. S.L. Sutton, T.C. Whitemore & A.C. Whitemore), pp 377-384.Blackwell, Oxford.

Low, A.B. & Rebelo, A. (1996). *Vegetation of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs and Tourism, Pretoria.

Margules, C.R., Nicholls, A.O. & Pressey, R.L. (1988). Selecting networks of reserves to maximise biological diversity. *Biological Conservation* 43, 63-76.

Margules, C.R., Cresswell, I.D. & Nicholls, A.O. (1994). A scientific basis for establishing networks of protected areas. In *Systematics and Conservation Evaluation*. (Eds. P.L. Forey, C.J. Humphries & R.I. Vane-Wright), pp 327-350. Clarendon Press, Oxford.

Macdonald, I.A.W. (1989). Man's role in changing the face of southern Africa. In *Biotic diversity in Southern Africa. Concepts and conservation*. (Ed. B.J. Huntley), pp 51-77. Oxford University Press, Cape Town.

McFarlane, D.J., George, R.J. & Farrington, P. (1993). Changes in the hydrological cycle. In *Reintegrating fragmented landscapes: Towards sustainable production and nature conservation* (Eds. R.J. Hobbs & D.A. Saunders). Springer-Verslag, New York.

Mugo, D.N., Lombard, A.T., Bronner, G.N. & Gelderblom, C.M. (1995). Distribution and protection of endemic or threatened rodents, lagomorphs and macrosceledids in South Africa. *South African Journal of Zoology* 30, 115-127.

Muller, C. (1999). *The distribution and conservation of termites (Isoptera) and butterflies (Lepidoptera) in South Africa*. Unpublished MSc dissertation. University of Pretoria, Pretoria, South Africa.

Nantel, P., Bouchard, A., Brouillet, L. & Hay, S. (1998). Selection of areas for protecting rare plants with integration of land-use conflicts: A case study for the west coast of Newfoundland, Canada. *Biological Conservation* 84, 223-234.

Nicholls, A.O. & Margules, C.R. (1993). An upgraded reserve selection algorithm. *Biological Conservation* 64, 165-169.

Noss, R.F. & Harris, L.D. (1986). Nodes, networks and MUM's: preserving diversity at all scales. *Environmental Management* 10, 299-309.

Nulsen, R.A. (1993). Changes in soil properties. In *Reintegrating fragmented landscapes: Towards sustainable production and nature conservation*. (Eds. R.J. Hobbs & D.A. Saunders), pp 107-145. Springer-Verslag, New York.

O'Connor, T.G. (1991). Local extinction in perennial grasslands: a life-history approach. *American Naturalist* 137, 735-773.

Palgrave, KC. (1983). *Trees of Southern Africa*. Struik Publishers, Cape Town.

Pimentel, D., Stachow, U., Takacs, D.A., Brubaker, H.W., Dumas, A.R., Meaney, J.J., O'Neil, J.A.S., Onsi, D.E. & Corzilius, D.B. (1992). Conserving biological diversity in agricultural-forestry systems. *BioScience* 42, 354-362.

Pimm, S.L. & Lawton, J.H. (1998). Planning for Biodiversity. *Science* 279, 2068-2069.

Poore, D. (1978). Ecosystem conservation. In: *Sourcebook for a world conservation strategy:* 25. Morges, Switserland: General assembly paper GA78/10 add 4 IUCN.

Pressey, R.L. (1992). Nature conservation in rangelands: Lessons from research on reserve selection in New South Wales. *Rangelands Journal* 14, 214-226.

Pressey, R.L. (1994). *Ad hoc* reservation, Forward or backward steps in developing representative reserve systems? *Conservation Biology* 8, 662-668.

Pressey, R.L., Ferrier, S., Hager, T.C., Woods, C.A., Tully, S.L. & Weinman, K.M. (1996). How well protected are the forests of north-eastern New South Wales? – analyses of forest environments in relation to formal protection measures, land tenure and vulnerability to clearing. *Forest & Ecological Management* 85, 311-333.

Pressey, R.L., Ferrier, S., Hutchinson, C.D., Sivertsen, D.P., & Manion, G. (1995). In *Nature Conservation 4, the role of networks.* (Eds. D.A. Saunders, J.L. Craig & E.M. Mattiske), pp 23-33. Surrey Beatty and Sons, Sydney.

Pressey, R.L., Humphries, C.L., Margules, C.R., Vane-Wright, R.I., & Williams, P.H. (1993). Beyond opportunism, Key principles for systematic reserve selection. *Trends in Ecology & Evolution*, 8, 124-128.

Pringle, E.L.L., Henning, G.A. & Ball, J.B. (1994). *Pennington's Butterflies of Southern Africa.* 2nd edition. Struik Winchester, Cape Town.

Pressey, R.L. & Logan, V.S. (1997). Inside looking out: Findings of research on reserve selection relevant to "off-reserve" nature conservation. In *Conservation Outside Nature Reserve.* (Eds. P. Hale & D. Lamb), pp 407-418. Centre for Conservation Biology, University of Queensland.

Rebelo, A.G. & Siegfried, W.R. (1992). Where should nature reserves be located in the Cape floristic region, South Africa? Models for the spatial configuration of a reserve network aimed at maximizing the protection of floral diversity. *Conservation Biology* 6, 243-252.

Saunders, D.A., Hobbs, R.J. & Arnold, G.W. (1993). The Kellerberrin project on fragmented landscapes: A review of current information. *Biological Conservation* 64, 185-192.

Scholtz, C.H. & Chown, S.L. (1993). Insect conservation and extensive agriculture: the savanna of southern Africa. In *Perspectives on Insect Conservation.* (Eds. K.J. Gaston, T.R. New & M.J. Samways), pp 75-95. Intercept, Andover.

Scotney, D.M., Botha, A.D.P., Cloete, J.G., Cowan, G., Dreyer, L. Koch, F.G.L, LeRoux, G.H., Little, A.M., Olivier, J. & Saunders, J. (1988). Agricultural areas of southern Africa. In *Long-term data series relating to southern Africa's renewable natural resources. South African National Scientific Programmes Report no. 157*. (Eds. I.A.W. Macdonald & R.J.M Crawford), pp 316-336. CSRI, Pretoria.

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, H., Caicco, S., D'Erchia, F., Edwards, T.C., Ulliman, J. & Wright, R.G. (1993). GAP Analysis: A Geographic approach to protection of Biological Diversity. *Wildlife Monographs* 123, 1-41.

Seymour, C. (1998). *Different grazing intensities in arid rangelands: Effects on invertebrates on a communal farm in the succulent Karoo, South Africa*. Unpublished MSc dissertation. University of Cape Town, Cape Town, South Africa.

Skelton, P.H., Cambray, J.A., Lombard, A.T. & Benn, G.A. (1995). Patterns of distribution and conservation status of freshwater fishes in South Africa. *South African Journal of Zoology* 30, 71-82.

Skinner, D.J. & Smithers, R.H.N. (1990). *The Mammals of the Southern African Subregion*. University of Pretoria, Pretoria.

Skinner, J.D. (*In Press*). Appendix: Red data book status. In *Mammals of the Southern African Subregion*, (3rd edn): University of Pretoria, Pretoria.

Smith, N.J.H. (1996). Effects of land-use systems on the use and conservation of biodiversity. In *Biodiversity and Agricultural Intensification*. (Eds. J.P. Srivastava, N.J.H Smith & D.A. Forno), pp 52-79. The World Bank, Washington, DC.

Soulé, M. S. (1991). Conservation: Tactics for a Constant Crisis. *Science* 253, 744-750.

Soulé, M. S. & Sanjayan, M.A. (1998). Conservation targets: Do they help? *Science* 279, 2060-2061.

Srivastava, J.P., Smith N.J.H. & Forno, D.A. (1996a). Agriculture as friend or foe of biodiversity. In *Biodiversity and Agricultural Intensification*. (Eds. J.P. Srivastava, N.J.H Smith & D.A. Forno), pp 1-10. The World Bank, Washington, DC.

Srivastava, J.P., Smith, N.J.H. & Forno, D.A. (1996b). Towards a strategy for mainstreaming biodiversity in agricultural development. In *Biodiversity and Agricultural Intensification*. (Eds. J.P. Srivastava, N.J.H Smith & D.A. Forno), pp 121-128. The World Bank, Washington, DC.

Thompson, M. (1996). A standard land-cover classification scheme for remote sensing applications in South Africa. *South African Journal of Science* 92, 34-42.

Todd, S. & Hoffman, M.T. (*In Press*). The effects of heavy grazing on plant species diversity and community composition in a communally managed, semi-arid shrubland, Namaqualand, South Africa. *Plant Ecology*.

UNEP. (1995). *Global Biodiversity Assessment*. Cambridge University Press.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S. & Mansell, M. (1998a). Biodiversity Assessment and Conservation Strategies. *Science* 279, 2106-2108.

van Jaarsveld, A.S., Ferguson, J.W.H. & Bredenkamp, G.J. (1998b). The Groenvaly grassland fragmentation experiment: design and initiation. *Agricultural Ecosystems & Environment* 68, 139-150.

van Jaarsveld, A.S., Gaston, K.J., Chown, S.L. & Freitag, S. (1998c). Throwing biodiversity out with the binary data? *South African Journal of Science* 94, 1-5.

Vandermeer, J. & Perfecto, I. (1997). The Agroecosystem: A need for the conservation biologist's lens. *Conservation Biology* 11, 591-592.

WCMC. (1992). *Development of a National Biodiversity Index: a discussion paper prepared by World Conservation Monitoring Centre. Report on the WCMC*, 15 September, 1992.

Western, D. (1989). Conservation without parks: wildlife in rural landscape. In *Conservation for the Twenty-first Century*. (Eds. D. Western & M. Pearl), pp 158-165. Oxford University Press, New York.

Wessels, K.J., Freitag, S. & van Jaarsveld, A.S. (1999). The use of land facets as biodiversity surrogates during reserve selection at a local scale. *Biological Conservation* 89, 21-38.

White, D., Minotti, P.G., Barczak, M.J., Sifneos, J.C., Freemark, K.E., Santelmann, M.V., Steinitz, C.F., Kiester, A.R. & Preston, E.M. (1997). Assessing risk to biodiversity from future landscape change. *Conservation Biology* 11, 349-360.

Wilson, E.O. (1988). The current state of biological diversity. In *Biodiversity*. (Ed. E.O. Wilson), pp 3-20. National Academic Press, Washington.

**Appendix I:** Species which only occur in grid cells that are more than 50% modified or transformed.

| **Species** | **Common name** | **Status** | **Comments** |
|---|---|---|---|
| **Butterflies** | | | |
| *Hyalites cerasa* (previously *Acraea cerasa*) | Tree-top Acraea | Common elsewhere | Coastal forest species and forests of Mozambique. |
| Alaena margaritacea | Wolkberg Zulu | Vulnerable[1] | Confined to vicinity of Wolkberg mountains in Northern province. |
| *Antanartia hippomene* | Southern Short-tailed Admiral | Common elsewhere | Common to woodlands and forests south of study area. |
| *Cnodontes pallida* | Pale Buff | Common elsewhere | Very rare in S.A., common to Botswana and northern Namibia. |
| *Coeliades anchises* | One-pip Policeman | Uncommon | Occurs in bushveld region of study area. |
| *Deudorix penningtoni* | Pennington's Playboy | Common | Found in a few localities within Mpumalanga and Northern province. |
| *Lepidochrysops letsea* | Free State Blue | Common elsewhere | Occasionally recorded in Gauteng. |
| *Neptis alta* | Old Sailer | Common elsewhere | Only a few known records south of Limpopo river, i.e. S.A. |
| *Neptis kiriakoffi* | Kiriakoff's Sailer | Common elsewhere | Very rarely recorded in South Africa, but common in Mozambique and Zimbabwe |
| *Spialia agylla* | Grassveld Sandman | Common | Wide range throughout southern Africa, including Gauteng. |
| *Stygionympha robertsoni* | Robertson's Brown | Common elsewhere | Rarely recorded in study area, common throughout most of the arid south-western Africa. |
| *Stygionympha vigilans* | Western Hillside Brown | Common elsewhere | Rarely recorded in study area, common along mountain ranges of south-western Cape of S.A. |
| **Mammals** | | | |
| *Eidolon helvum* | Straw coloured fruit bat | Uncommon | Migrant of tropical African forests. |
| *Scotophilus nigrita* | Giant yellow house bat | Uncommon | Very rare throughout Africa. |
| *Grammomys cometes* | Mozambique woodland mouse | Uncommon | Widespread through Africa, also found in south-eastern Mpumalanga and northern Kwazulu-Natal. |
| **Birds** | | | |
| Sarothrura affini | Stripped flufftail | Threatened[2] | Occurs in montane grassland of Mpumalanga. |
| *Cursorius rufus* | Burchell's courser | Common elsewhere | Common to dry western region of southern Africa. |
| *Turtur afer* | Bluespotted dove | Uncommon | Occurs in evergreen and riverine forests. |

| | | | |
|---|---|---|---|
| *Motacilla cinerea* | Grey Wagtail | Uncommon | Palearctic migrant, non-breeding visitor to Africa. |
| **Plants (Endemic, i.e. within South Africa only recorded in study area.)** | | | |
| *Aloe alooides* (Bolus) | Graskop aloe | Locally common | Common in inaccessible mountains of Mpumalanga. |
| *Aloe lutescens* | Aloe family | Uncommon | Found between Soutpansberg and Limpopo river. |
| *Aloe marlothii subsp. marlothii* | Mountain aloe | Uncommon | Found in Gauteng, Pretoria, Magaliesberg, Suikerbosrand. |
| *Aloe parvibracteata* | Aloe family | Uncommon | Occurs in Mpumalanga, but also possibly in Kwazulu-Natal. |
| *Aloe peglerae* | Turk's cap, Mountain aloe, Red hot poker | Rare[3] | Rare and confined to Magaliesberg and Witwatersberg in Gauteng. |
| *Blechnum australe var. australe* | Fern | Uncertain | Also recorded elsewhere in Africa, i.e. Zimbabwe, Kenya. |
| *Blechnum sp.* | Fern | Uncertain | Undescribed species of cosmopolitan genus with six species in S.A. and three varieties endemic to eastern parts of subcontinent. |
| *Borassus aethiopum* | Borassus Palm. | Rare[3] | Rare and protected in Northern province, but also found north of Limpopo river. |
| *Cheilanthes inaequalis var. inaequalis* | Ferns and fern allies | Uncertain | Found in north-eastern parts of S.A., but also elsewhere in Africa. |
| *Cyperus elephantinus* | Cyperaceae family, Sedge family | Uncertain | Occurs in Northern province and tropical Africa. |
| *Cyperus fulgens var. contractus* | Cyperaceae family, Sedge family | Uncertain | Occurs in Northern province and tropical Africa. |
| *Dryopteris athamantica* | Pannae-radix | Uncertain | Eastern parts of Southern Africa and tropical Africa |
| *Eriocaulon sp.* | Pipewort family | Uncertain | Undescribed species, possibly also occurs elsewhere in wet parts of S.A. |
| *Marsilea capensis* | Fern | Uncertain | Widespread in Africa, i.e. Zambia and Egypt. |
| *Scirpus ficinioides* | Cyperaceae family, Sedge family | Uncertain | Found in Mpumalanga and Gauteng, but also elsewhere in Africa. |

1. South African Red Data Book – Butterflies, Henning & Henning (1989).
2. South African Red Data Book – Birds, Brooke. (1984).
3. Red Data List of Southern African Plants, Hilton-Taylor (1996).