The Application of Systematic Conservation Planning in the Succulent Karoo Biome of South Africa.

by

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Dedication

This thesis is dedicated to the memory of Leslie Hill

Abstract

Designing a Living Landscape for Biodiversity Conservation in the Knersvlakte Region of the Succulent Karoo, South Africa: A Systematic Conservation Planning Approach

by Philip George Desmet *February 2004*

Systematic conservation planning is about making spatially explicit decisions regarding the use of land, based on the observed or expected biodiversity present at a site and the potential for that same site to support alternative land-uses that are not compatible with the persistence of biodiversity. This thesis examines three questions relating to the application of systematic conservation planning: Which biodiversity surrogates should be used in Namaqualand to do systematic conservation plans? How should targets be set for these surrogates? How can this information be integrated and used within a systematic conservation planning framework?

Comparing how well different biodiversity surrogates achieved a set of targets illustrated that continuous biodiversity data (i.e. vegetation types and land-classes) perform better as surrogates than point-based species distribution data. Quarter degree square-based species distribution data cannot be used for on-the-ground conservation planning.

It was demonstrated that it is possible to set biologically meaningful conservation targets to represent biodiversity pattern in land classes by applying the Species Area Relationship and using plot-based survey data. The method developed here has the potential to revolutionise conservation planning as it provides for the first time a defensible means for setting representation targets for land classes that are grounded on ecological theory and that use real data.

The thesis also explores the potential for metapopulation and fragmentation studies to provide useful insights into developing targets for ecological processes by relating the amount of remaining habitat to key thresholds in probability of population persistence.

Two examples, at different spatial scales (1:10 000 and 1:100 000), are used to illustrate how different biodiversity information can be integrated and used within a systematic conservation planning framework. At the finer scale biodiversity and land-use data are

used to set priorities for the development of a statutory reserve in the Knersvlakte region of the Succulent Karoo using cadastres as planning units. At the larger scale the data are used in the same region to design a biosphere reserve that promotes the persistence of ecological processes in the landscape using gridded planning units. Both studies use the C-Plan software to assist in the planning and design process. A lesson from both these studies is that there needs to be a paradigm shift in conservation from an on/off reserve mindset to a more integrative whole landscape mindset.

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1 Introduction

Conservation planning is a branch of conservation biology that seeks to identify spatially explicit options for the preservation of biodiversity (Pressey *et al.* 1993; Williams *et al.* 1996). It involves making decisions about the use of a parcel of land based on the biological, environmental and anthropogenic attributes of that parcel and its neighbours. Alternative systems of conservation areas are, in essence, hypotheses about effective ways of promoting the persistence of biodiversity. Invariably, these options are constrained by a number of factors, such as the existing reserve system (Pressey 1994a), the extent and configuration of transformed habitat (Lombard *et al.* 1997a), and forms of land use that are financially more viable (at least in the short term) than conservation (Ferrier *et al.* 2000).

To be most effective, conservation planning should be systematic. Systematic approaches share the following features: they are data driven; target directed; efficient; transparent and repeatable; and flexible (Cowling *et al.* 1999a; Pressey 1999; Margules and Pressey 2000a). One of the important prerequisites is that the data used in making these decisions are spatially explicit. Biological features (e.g. species, subspecies, "evolutionary significant units", "management units", habitats, landscape units, etc.) and their patterns of occurrence (e.g. range sizes, extent of suitable habitat, migration patterns, etc.), that act as surrogate measures of biodiversity, need to be identified in precise terms if they are to be targeted for conservation action.

While a systematic planning approach based only on the distribution of biological features in a landscape may efficiently identify a set of conservation priorities, it has a major limitation. The outcome reflects the options for achieving targets for biodiversity pattern only. Reserve systems that are designed to retain only biodiversity pattern will not ensure long-term conservation. This is because these systems do not explicitly consider the ecological and evolutionary processes that maintain and generate biodiversity (Frankel and Soule 1981; Hunter *et al.* 1988; Moritz 1994; Balmford *et al.* 1998; Cowling *et al.* 1999a).

Thus, systematic conservation planning comprises two key activities. Firstly, identifying the biodiversity pattern and process features that one wants to include in a conservation plan; and, secondly, setting quantitative conservation targets for these features. A third key component of systematic conservation planning is then putting this information together and producing a plan. These three components are essentially the core of the initial steps in planning and they are the focus of this thesis. Naturally, taking a plan forward to implementation is a whole different game that is not the primary focus of this thesis.

This thesis asks three questions:

1. Which biodiversity surrogates should be used in Namaqualand to do systematic conservation plans?
2. How should targets be set for these surrogates?
3. How can this information be integrated and used within a systematic conservation planning framework?

In this thesis I set about exploring these questions by drawing on my work from the numerous projects I have been involved with over the last five years. The real world context for all of this research present here is the phenomenally diverse Succulent Karoo biome of South Africa.

There are no true measures of "biodiversity". Invariably biodiversity is always quantified through some form of surrogate. The choice of which biodiversity surrogate to use in planning is certainly topical in the conservation literature. I have chosen to begin the thesis by discussing the utility of biodiversity data typically available to conservation planners here in South Africa. In Chapter 2 I compare point and quarter degree square species distribution to expert derived and modelled vegetation maps. There is no right or wrong biodiversity surrogate data, but some kinds are more useful than others in the planning context.

Setting biologically meaningful conservation targets is probably the biggest challenge facing conservation planners. Without a sound biological basis for target setting it is exceedingly difficult to justify these targets. Chapter 3 presents a method for setting vegetation type targets using relevé data and the species-area relationship (SAR). Since the 1970's ecologist have applied the SAR to a range of conservation and landscape ecological questions; however, no one has ever explicitly linked it with setting conservation targets. A distilled version of this chapter was presented at the World Parks Congress in Durban in 2003 and it is currently in press in *Ecological Conservation*.

Chapter 4 takes the target issue further by attempting to apply landscape ecology and meta-population theory to addressing the even more challenging question of setting targets for ecological processes. This Chapter reviews relevant literature and draws out some useful conclusions that are applied in designing the Knersvlakte Biosphere Reserve in Chapter 6.

The remaining two Chapters of the thesis demonstrate how the theory on systematic conservation planning can be applied in the real world. Both use the extraordinary Knersvlakte region of the Succulent Karoo as the planning domain. Chapter 5 focuses on the design of a single statutory reserve in the region focussed on conserving a representative sample of the unique biodiversity attributes of the region, namely the quartz patches, limestone and quartzite rock habitats. This study uses individual land parcels as planning units and focuses on identifying specific properties that can be incorporated into a formal reserve.

Chapter 6 zooms out to look at the whole landscape and focuses on identifying the spatial requirements for ecological processes necessary to maintain the biodiversity of the core reserve as well as that of the broader landscape as a whole. Here the planning units are grids and the product does not identify individual cadastres but rather presents a biologically relevant spatial framework for the development of a biosphere reserve in the region. The two studies use very similar biodiversity data; however, these data are used to address conservation planning questions at two very different spatial and implementation scales – reserve design and land-use planning.

The major goal of these two chapters is to demonstrate, step-by-step, how to actually do conservation planning. Consequently these chapters flow as narratives of the planning process rather than research chapters focussed on a particular question. Examples of how one goes about doing a conservation plan is rarely published in the literature and I decided early on that this thesis would be aimed at assisting conservation practitioners in South Africa in this regard. This thesis is also aimed at assisting the Leslie Hill Succulent

Karoo Trust in prioritising the purchase of conservation worthy land in the Knersvlakte , which provided the impetus for much of the work contained in this thesis.

2 The Use of Biodiversity Surrogates in Regional Conservation Plans

2.1 Introduction

The goal of systematic conservation planning is to plan for the representation and long-term persistence of biodiversity within an area or region. For this goal to be achieved, all facets of biodiversity must be represented spatially if biodiversity is to be targeted effectively in the planning process (Cowling *et al.* 1999a; Margules and Pressey 2000b; Pressey *et al.* 2003b). As there are few true measures of biodiversity (Sarkar and Margules 2002), the task of representing biodiversity relies on the use of biodiversity estimators or surrogates to reasonably reflect the underlying biological patterns and processes that together comprise "biodiversity". A challenge facing conservation planning is deciding what constitute reasonable surrogates for biodiversity. Perhaps a bigger challenge lies in determining how these surrogates can be sufficiently well mapped as to provide useful input for the conservation planning process. All conservation plans invariably use incomplete data on biodiversity pattern (Ferrier 2002). Thus, it is important to understand how well different features perform as biodiversity surrogates.

This chapter examines the role that two most common classes of biodiversity surrogates, namely species distribution data and land-class maps, can play in the conservation planning process. It does not address the traditional biodiversity surrogate debate of comparing the congruence between surrogates, but rather focus on how the results of such analyses inform us as to the utility and limitations of different biodiversity surrogates.

The data needed to prioritize areas for biodiversity protection are records of biodiversity features such as species, species assemblages, or environmental classes for each candidate area (Williams *et al.* 2002). Biodiversity surrogates for biodiversity pattern generally fall into two broad categories: (a) discrete taxonomic distribution data; or, (b) continuous land-class data. Taxonomic data may comprise point or grid-square based distribution records for species or genera derived from museum records or dedicated taxonomic surveys. Categorical land-class data typically comprises continuously mapped higher-order biodiversity surrogates such as vegetation community types or environmental classes derived from expert mapping or some form of GIS-based modeling exercise, or a combination of the two.

A common approach in regional conservation assessments is to use data on recorded species as surrogates for unsampled species diversity (Rebelo and Siegfried 1992b; Beccaloni and Gaston 1995; Lombard *et al.* 1995; Freitag *et al.* 1998; Pearson and Carroll 1998; Ferrier *et al.* 1999; Lombard *et al.* 1999a). Determining how well surrogates perform as estimators for true biodiversity is not possible as it is not yet know how to accurately quantify biodiversity (Sarkar and Margules 2002). Therefore, the traditional approach to testing the appropriateness of surrogates is to determine how well the patterns of richness and endemism observed in the distribution in one surrogate approximate these patterns in other surrogates. An alternative approach is to examine how well achieving conservation targets for one surrogate achieve targets for other surrogates.

Tests comparing taxonomic surrogates have produced mixed results. Some studies show good congruence between areas selected for different taxa (Csuti *et al.* 1997; Howard *et al.* 1998; Ferrier *et al.* 1999), whereas other studies have shown a lack of congruence among the distributions of different taxa or areas selected to represent them (Prendergast *et al.* 1993; Lombard *et al.* 1995; Dobson 1997; Flather *et al.* 1997; Kerr 1997; Reid 1998; van Jaarsveld *et al.* 1998; Ferrier *et al.* 1999). Despite the propensity for many studies to use taxonomic data, there are numerous problems associated with using these data including incompleteness and spatial biases in species records, and biases toward species that are easy to observe or those for which the taxonomy is well established (Belbin 1993; Pressey 1994b; Faith and Walker 1996a; Faith 1996; Haila and Margules 1996; Noss 1996b; Lawes and Piper 1998; Margules and Pressey 2000b; Pressey *et al.* 2000; Ferrier 2002; Williams *et al.* 2002)

Many of these problems can be overcome by using remotely derived environmental data to model the distribution of species (e.g. Araujo and Williams 2000; Pearce and Ferrier 2000; Pearce *et al.* 2001; Rouget *et al.* 2001; Austin 2002; Bailey *et al.* 2002; Ferrier *et al.* 2002b) or communities of species (e.g. Franklin 1995; Cawsey *et al.* 2002; Ferrier

2002; Ferrier *et al.* 2002a), or environmental classes (e.g. Margules and Redhead 1995; Roy and Tomar 2000; Faith *et al.* 2001) as biodiversity surrogates. The environmental data that form the basis of the modeling approach are relatively easy to collect or interpolate, and are often the only option in data-poor areas or the world (Faith *et al.* 2001). Modeled approaches are inherently limited by the resolution and accuracy of the underlying interpolated environmental data. Also, models are poor at predicting distribution patterns that arise as a result of historical factors, such as the small-scale vicariance observed in the Cape and Succulent Karoo floras (Cowling and Lombard 2002b).

The alternative to these modeled approaches is to use expert input to map vegetation or other biologically meaningful categories directly in the field or from aerial imagery (Ferrier 2002). In addition to addressing problems of raw taxonomic data, the main advantage of using a continuous biodiversity surrogate, modeled or mapped, is that all areas of the landscape have some biodiversity information attached to it. In contrast when using discrete species distribution records, no decisions can be made about areas that have no data. Unless the absence of data reflects true absences, these areas are effectively invisible to the conservation planning software.

Whichever approach is taken to mapping biodiversity, it is still necessary to show that the surrogates are representative of species or communities in general before they can be used with confidence in conservation plans (Pressey 1994b; Reyers and van Jaarsveld 2000; Araujo and Williams 2001). As discussed above, with taxonomic data the approach is often to compare how well one species group reflects patterns in other species groups. A similar approach is often taken to testing vegetation types or land-classes (Kiester *et al.* 1996; Ferrier and Watson 1997; Ferrier 2002). Studies conducted in South Africa have shown that land classes are generally good surrogates for a variety of animal and plant taxonomic groups (Wessels *et al.* 1999; Reyers *et al.* 2002; Lombard *et al.* 2003). However, the scale of environmental unit mapping is important with more finely delineated land-classes better reflecting underlying taxonomic patterns (Araujo *et al.* 2001; Reyers *et al.* 2002). A better approach would be to use community data directly in the modeling process to help delineate the mapped environmental classes. This would circumvent this problem of demonstrating surrogacy (Ferrier 2002).

Given the biodiversity surrogate options available to conservation practitioners, this chapter focuses on the use of such data in regional conservation assessments by addressing the following two questions:

1. How can species locality data be used effectively in regional conservation assessments?
2. What are the limitations of using species data, or land-class data, alone in regional conservation assessments?

The analyses use real data from the arid Namaqualand region of northwestern South Africa. The analyses compare expert mapped vegetation types, modeled land-classes, and, herbarium (grid-based) and survey plant (point locality) species distribution data against one another to test how well each achieves, either alone or together, targets for the other surrogates. These analyses are performed at two spatial scales reflecting the two scales most commonly used for conservation planning in the sub-continent. These are by no means novel research questions, but for the purposes of planning in the Succulent Karoo they serve to illustrate the limitations of the data available for planning.

Comparisons are made using two approaches. Firstly, comparing the probability that achieving the targets for one feature will achieve the targets for another using a modification of the irreplaceability-based method presented by Lombard *et al.* (2003). This method is effectively the same as performing a suite of minsets to determine the probability of an area being conserved based on the surrogate data (e.g. Hopkinson *et al.* 2001). This novel method provides a computationally efficient means of comparing surrogates for large datasets as well as a means of dealing with the inflexibility of single minset outcomes. The method is also able to deal with presence/absence type taxonomic data (e.g. a species is either recorded at a site or not) as well as aerial type land-class data (e.g. 2000 ha of a vegetation type at a site). The original method of Lombard *et al.* (2003) could only deal with taxonomic type data and not land-class type data.

The analyses are performed within a framework free from the influence that confounding factors such as the existing reserve network and patterns of habitat transformation would have on the results. The objective of this paper is to examine the potential role of biodiversity surrogates in regional conservation assessments, not make real-world recommendations about which areas to conserve. Normally it would be expect that habitat transformation would influence the distribution of biodiversity and hence conservation outcomes; however, these are context specific and including this here would confound interpretation of the results.

The second approach compares minset outcomes by calculating the number of additional sites required to achieve targets for other surrogates after the minset has already achieved the targets for one or more surrogates. For an ideal biodiversity surrogate no additional sites would need to be added meaning that achieving targets for the ideal surrogate also achieves targets for all other biodiversity surrogates used in the analysis.

2.2 *Methods*

2.2.1 Study area

The study area (planning domain) is located in the northwest corner of South Africa with Namibia (Figure 2.1). The area is bounded in the north by the Gariep River, which is the border between the two countries, and in the south and southeast by the Olifants River and Bokkeveld escarpment. This region of the country is known as Namaqualand. The vegetation is predominately succulent karoo with thicket and fynbos vegetation on the higher mountains (Cowling *et al.* 1999b; Dean and Milton 1999). This area is part of the Succulent Karoo Biome, which forms the arid winter-rainfall fringe of the Cape Floral Kingdom. The Succulent Karoo is rich in endemic plant taxa (168 families, 1002 genera and 6356 species with 26% of taxa strict endemics), and composed of predominately succulents and geophytes. Unlike the neighboring Fynbos Biome (Lombard *et al.* 2003), the Succulent Karoo is relatively data poor when it comes to point locality biodiversity data for any taxonomic group.

2.2.2 Spatial scales

Analyses were performed at two spatial scales. The coarser-scale quarter degree squares (QDS, Table 2.1) are the resolution at which all herbarium collection data, as well as other museum taxonomic data, in South Africa are geo-referenced. This is also the scale at which most regional conservation prioritization studies have operated (e.g. Lombard *et al.* 1995; Freitag and VanJaarsveld 1997; Muller *et al.* 1997; Lombard *et al.* 1999b; Reyers *et al.* 2002).

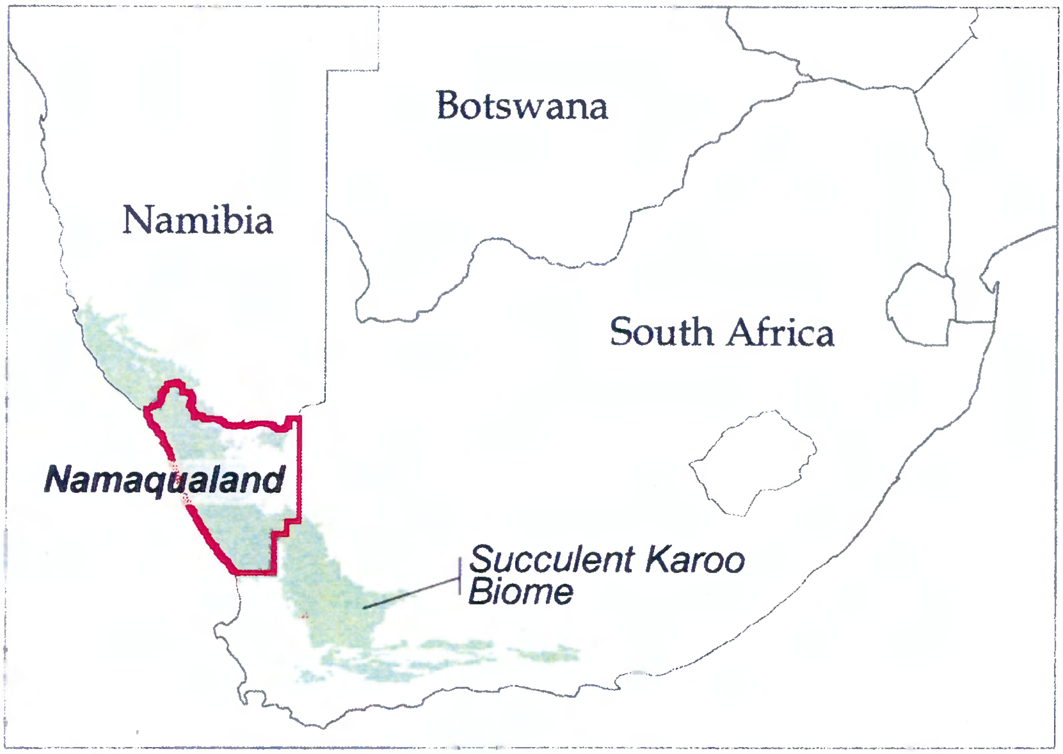


Figure 2.1: Location of the study area.

The finer-scale sixteenth degree square (SDS, Table 2.1) represent a much smaller planning unit used in two applied conservation planning projects recently undertaken in South Africa (Cowling *et al.* 2003c; Driver *et al.* 2003b). Whilst this planning unit size is more applicable to regional planning exercises, very few taxonomic datasets can be resolved to this scale unless the data were originally collected as point distribution data (e.g. Lombard *et al.* 2003). The implications for data used in the planning process are discussed in the following section.

Table 2.1: The two spatial scales at which analyses were performed.

Planning Unit	Number of Units in Planning Domain	Planning Unit Dimensions (lat. x long.)	
		Degrees	Kilometers
1. Quarter degree squares	120	15′ x 15′	24 x 27
2. Sixteenth degree squares	1712	3.57′ x 3.37′	6 x 7

2.2.3 Biodiversity features

2.2.3.1 Vegetation Types

A vegetation map is a continuous biodiversity surrogate information layer. It often comprises vegetation types based on the interpretation of landscape-scale vegetation patterns by relevant botanical experts. The vegetation map used in this study is based on the new South African vegetation type map (National Botanical Institute, Pretoria). The map used here was modified for the Succulent Karoo Ecosystem Plan (SKEP) project by the inclusion of some azonal vegetation types (e.g. quartz patches) (Figure 2.2) (Driver *et al.* 2003b). There are a total 100 SKEP vegetation types in the study area.

2.2.3.2 Land-Classes

The land-class map acts as another type of continuous biodiversity surrogate similar to vegetation types (Figure 2.2). This map was developed for Namaqualand in conjunction with Simon Ferrier and Glen Manion form the New South Wales Parks and Wildlife Service in Armidale, Australia (Ferrier *et al.* in prep.). The method used a generalised additive model to combine indices of soil and climate, and then to classify all possible combinations into biologically meaningful classes based on a cluster analysis of vegetation community data that was compiled for the region. All input environmental data layers were in ArcInfo grids at a 100m grid-cell size resolution. The vegetation community database comprised a collection of 5567 phytosociological relevé samples from across Namaqualand. There are a total of 94 land-classes in the study area.

2.2.3.3 *Conophytum* distribution data

The genus *Conophytum* (Aizoaceae) comprises small to extremely small leaf-succulent plants that are characteristic of the strongly winter-rainfall (i.e. western) Succulent Karoo (Figure 2.3). There are 170 recognized taxa in the genus distributed throughout the Succulent Karoo from Steytlerville in the east to Luderitz in Namibia. The *Conophytum* database represents the only taxonomically correct, species-level point-distribution database available for Namaqualand. The database comprises collections and observation records derived from expert botanists spanning the last 60 years of botanical exploration in the Succulent Karoo (Desmet and Hammer in prep.).

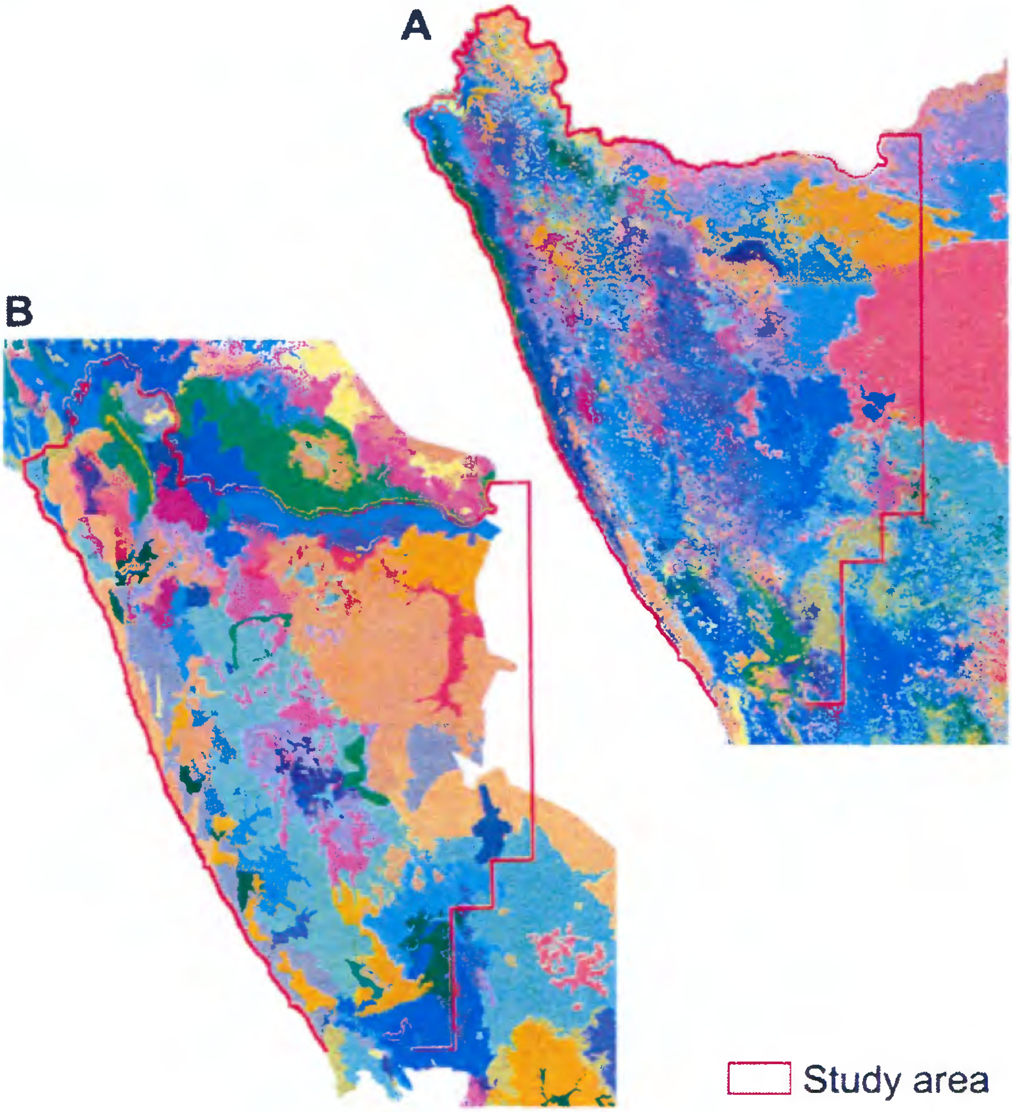


Figure 2.2 The distribution of (a) modelled land-classes and (b) expert mapped vegetation types in the study area.

There are 1249 individual point records from the study area. At the coarse scale, there are 504 unique species by QDS records (i.e. duplicate species records per QDS are excluded) with a total of 82 of the 120 QDS sites in the study area having occurrence data. At the finer spatial scale, there are a total of 883 unique species by SDS records with only 368 of the 1712 SDS sites having data. There are 135 *Conophytum* taxa represented in the study area out of a total of 170 taxa in the whole genus. The distribution of data points demonstrates a bias towards well-collected area such as known centers of diversity and near roads. Also, conophytums occur only on rocky substrata and therefore do not occur in areas of with extensive sandy substrates.

Figure 2.3 Some examples of conophytums (clockwise from top left): *Conophytum ernstii* flowering in autumn (Picture courtesy of Tom Jacobs). A natural hybrid between *C. bilobum* and *C. ectypum* from the Springbok area showing grotesque hybrid vigor. Orange flowers only occur in hybrid taxa in *Conophytum*. The very fuzzy *Conophytum mirabile* in it's deeply shaded natural habitat. This species total range is an area no larger than a tennis court. *Conophytum minutum* var. *pearsonii* is a widespread species from the southern Kamiesberg. Note the seedling in the bottom right-hand corner of the pot. Note also the size of plants relative to the pot labels. The examples of plant sizes shown here are average to large for the genus.

2.2.3.4 Endemic plant herbarium records

The Succulent Karoo is rich in endemic taxa. A total of 1630 taxa or 26% of the Succulent Karoo's flora are strict endemics to the biome (Driver *et al.* 2003b). This figure was calculated from an analysis of herbarium record data from South Africa (PRECIS South Africa and SABONET) and Namibia (PRECIS Namibia) and expert input for taxa not captured in any of these databases (Driver *et al.* 2003b). These herbarium data are georeferenced to the QDS scale.

The combined species database for the Succulent Karoo comprises 78 712 unique species by QDS occurrence records. Of the 1630 endemic taxa, 451 members of the Aizoaceae (ex. Mesembryanthemaceae) do not have QDS records, as specimens do not occur in any of the herbaria included in the above databases. Of the remaining 1179 taxa for which there are records, 721 occur in the study area in 96 of the 120 QDS. Of the remaining 24 QDS without endemic taxa records, only 6 do not have any herbarium records in the database.

2.2.4 Targets for features

Targets for each surrogate were developed only to represent species or biodiversity pattern. Process targets, such as multiple representation species, are not considered here. For each feature the targets were as follows:

1. For each vegetation type representation targets were set using the species-area relationship method discussed in Chapter 3. These targets range between 10% and 35% for the different vegetation types. This target is an estimate of the amount of area of a vegetation type that is required to represent 75% of the species that occur in that vegetation type at least once.
2. As the modeling process for land-classes controls for differences in patterns of species distribution across the landscape, a single target for land classes was set as the average of that for the vegetation types (20%). Thus, the total area of the study area targeted by vegetation types and land-classes is the same.
3. *Conophytum*: 1 occurrence of each taxon.
4. Endemic plant taxa: 1 occurrence of each taxon.

2.2.5 The final data sets

Combining the different site and surrogate datasets, two datasets were developed for analysis:

1. A QDS site by feature (surrogate) dataset containing 120 sites and 1050 features. The feature set included vegetation types, land-classes, conophytum and endemic plant distribution records.
2. A SDS site by feature (surrogate) dataset containing 1712 sites and 329 features. The feature set included vegetation types, land-classes and *Conophytum* distribution records.

2.2.6 Conservation planning software

We used C-Plan (Anon. 2001) to calculate site irreplaceability. C-Plan is a conservation-planning package that runs with the GIS software ArcView (ESRI, Redlands, California). C-Plan is used to identify a notional reserve system that will satisfy specified conservation targets for biodiversity features. Biodiversity features can be land classes, species or processes, and targets are set in either area units (e.g. hectares) for land classes and processes, or as numbers of occurrences of species (e.g. one occurrence of each) for species locality data sets.

C-Plan prioritizes sites based on a computed measure of conservation value, namely irreplaceability (Ferrier *et al.* 2000). The irreplaceability index is a measure assigned to a site that reflects the importance of that site, in the context of the planning domain, for achieving conservation targets for a given set of biological features. Site irreplaceability is a function of how much of each target is achieved. Thus irreplaceability can be viewed in two ways (Margules and Pressey 2000a; Anon. 2001):

- The potential contribution of any site to a conservation goal or the likelihood of that site being required to achieve the goal. In other words, the likelihood that a site will be selected by a minset to achieve a set of conservation targets.
- The extent to which the options for achieving a system of conservation areas, which is representative (i.e. achieves all the targets), are reduced if that site is lost or made unavailable.

2.2.7 Developing a method to compare surrogates

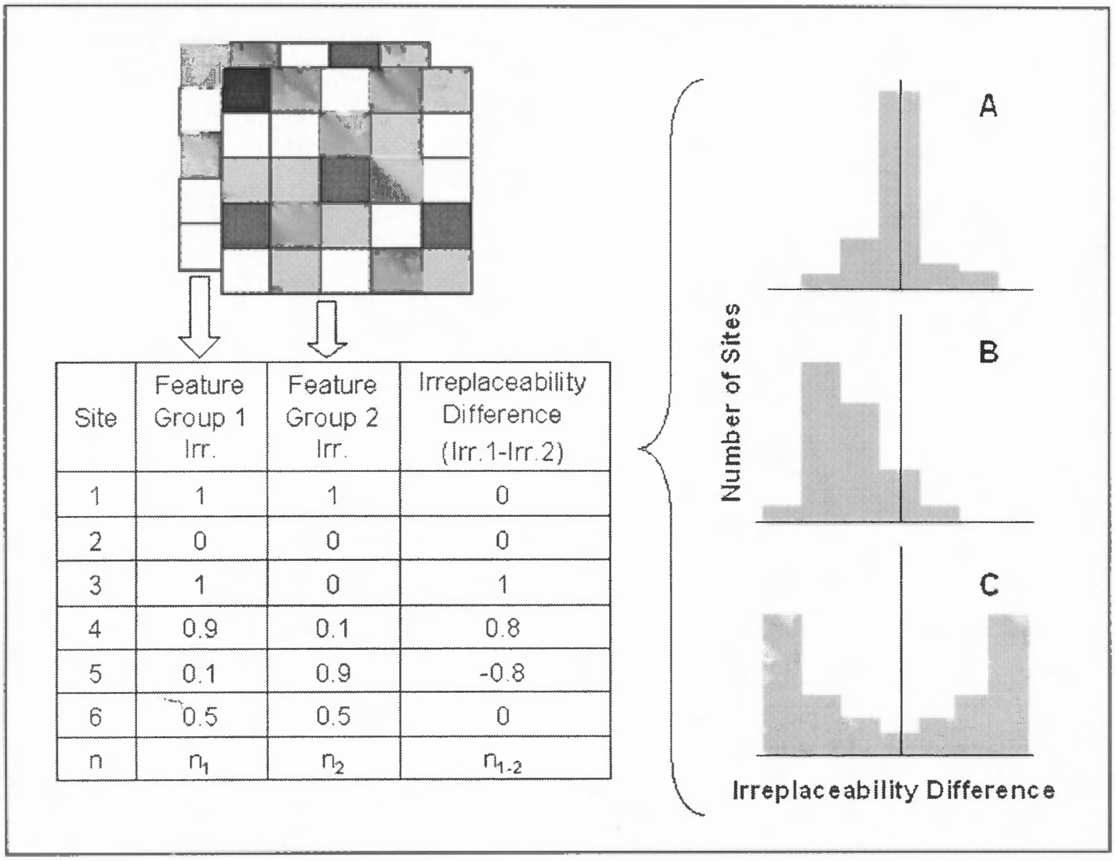
To demonstrate the utility of a biodiversity surrogate, it needs to be shown that achieving targets for one surrogate will achieve targets for one or more other surrogate groups. When selecting sites to meet targets for a surrogate, unless a feature is restricted to a single site there are spatial options for how these targets can be achieved in the landscape. As there are options as to which sites would be selected to achieve targets, a good way of estimating the probability of a site being selected to meet a target set would be to be to run a series of minsets, say 100 or 1000 times, to achieve the targets for a particular surrogate group, and then summarize for each selection unit the number of times it was selected. This would represent the probability of that site being required to achieve the target. This probability map could then be compared to see how well it achieves targets for other surrogate groups and determine what proportion of times a minset outcome for one feature group achieves the targets for another feature group. For an ideal surrogate each potential minset outcome should achieve most if not all targets for all of the comparison surrogates.

There is one major assumption regarding minset outcomes that only holds for these types of desktop analyses. In reality there can only ever be one true minset outcome as no two sites share the same variety and extent of biodiversity. In practice two sites can be equal in the planning context when using biodiversity surrogates. Given that there is more than one minset outcome, i.e. there are choices as to which sites are selected to achieve the targets set, testing the congruence between two surrogates requires that all possible outcomes are compared in order to determine the probability of one surrogate, say vegetation types, achieving the targets for another surrogates, for example reptile distribution.

Performing this type of analysis would require the development of dedicated software. C-Plan has the capacity to perform multiple minsets, however, there were several limitations with this approach. Firstly, despite performing the multiple minsets, C-Plan only records the results of the final outcome and none of previous iterations. Secondly, and more importantly, C-Plan has the limitation that it cannot randomly select a site from the list of available sites in the event of a tie being reached after all selection rules have been applied. This means that C-Plan will always reproduce that same outcome from a given rule-set even when several are possible due to some sites having identical features sets. Alternative software packages investigated (CODA, SITES and WORLDMAP) did not appear to have the functionality to perform multiple minset analyses.

Actually, there is no need to perform any minset analyses. Lombard *et al.* (2003) proposed an alternative solution that uses the site irreplaceability statistic calculated by C-Plan rather than performing multiple minsets. As irreplaceability is an estimate of the probability of a site being selected to achieve a given set of targets, it is in effect also an estimate of the probability of a site being selected by a minset (Lombard *et al.* 2003). For example, a site with an irreplaceability of 1 will always be selected by a minset whereas a site with an irreplaceability of 0 will never be selected and a site with an irreplaceability of 0.4 will be selected in 40% of the minset outcomes. When using a land-class or species surrogate to calculate irreplaceability and then comparing to other species surrogates, represented at a site by either presence or absence (0 or 1), multiplying each site's irreplaceability for all the sites where a given species in the comparison surrogate dataset occurs, and then summing the products gives an overall probability of that species target being met as a result of achieving the original land class or species targets. This method only holds where the comparison species target is one occurrence of that species. However, this method of analysis does not hold where the comparison species target is greater than one occurrence or the feature is represented at a site as an area, such as would be for land-classes. Thus, comparing how well taxonomic distribution data achieves targets for land-class surrogates cannot be estimated with this method.

A solution to the problem is to compare irreplaceability maps (Figure 2.4), as irreplaceability is by definition also the likelihood of a site being selected by minset. This negates the need to perform an iterative minset analysis. If one surrogate was a good surrogate for another then, and *vice versa*, the irreplaceability maps for both surrogates should look almost identical. In other words each site would have the same probability of being selected irrespective of the surrogate being used. If for any given site, the site irreplaceability for one surrogate were similar to that for another surrogate, then the difference between the two would be close to zero. Performing this calculation across all sites and then plotting the frequency distribution would provide a qualitative indication of the degree of congruence between two surrogate (Figure 2.4). In Figure 2.4 if two surrogates were good surrogates for each other then the distribution would be a sites and then plotting the frequency distribution would provide a qualitative indication of the degree of congruence between two surrogates (Figure 2.4). In Figure 2.4 if two surrogates were good surrogates for each other then the distribution would be a symmetrical normal curve centered on zero. If, however, one surrogate were a good surrogate for another, but not the reverse, then the frequency distribution would be skewed to the left or right, and this asymmetry would increase as the degree of surrogacy decreased. If neither were good surrogates for each other then the graph would resemble a u-shape (Figure 2.4c).



Site	Feature Group 1 Irr.	Feature Group 2 Irr.	Irreplaceability Difference (Irr.1-Irr.2)
1	1	1	0
2	0	0	0
3	1	0	1
4	0.9	0.1	0.8
5	0.1	0.9	-0.8
6	0.5	0.5	0
n	n_1	n_2	n_{1-2}

Figure 2.4: A method using site irreplaceability for comparing how achieving targets for one surrogate achieves targets for another. The two surrogates in histogram A are both good surrogates for one another as the distribution of values are symmetrical and centred on 0. Here, achieving targets for either surrogate will achieve most targets for the other. In histograms B, surrogate 2 is a better surrogate than 1 as the distribution is skewed to the left. In histogram C, surrogates are negatively correlated. A random pattern, however, would indicate that the surrogates bare no relationship to one another.

2.2.8 Minimum sets

Minsets are performed to estimate the total number of sites required to achieve each surrogates targets. C-Plan was used to perform the minset analyses. For all surrogates and at both spatial scales the only rule required to avoid ties being produced was "summed irreplaceability highest". However, variations of using different rules and starting conditions were investigated, but in all cases the total number of sites required to achieve targets were the same, only the sites and order in which they were selected differed. Thus, the minset is a convenient measure of how well one biodiversity surrogate acts as a surrogate for another by observing the number of additional sites required to meet targets when adding other surrogates to the feature set.

2.3 Results

2.3.1 QDS-scale

At the QDS scale, the irreplaceability maps for the continuous versus taxonomic surrogates are quite different (Figure 2.5). Both *Conophytums* and endemic species require nearly the whole study area, at least for where there are data, to achieve targets (sites with irreplaceability = 0 have no data). This means most sites that have data have at least one species that is restricted to that site. When looking at the sample minsets, in the *Conophytum* dataset there are 43 taxa out of 135 that are restricted to a single site (Table 2.2). Similarly for the endemic taxa dataset 75 of the 94 sites that have data are required to achieve targets indicating a high degree of site endemism (Table 2.2) at the QDS scale.

Fewer sites are required to achieve the land-class targets than for vegetation types, and these sites are more evenly dispersed across the study area (Figure 2.6). When adding land-classes to the vegetation types feature set, only an additional two sites are required to meet the land-class targets indicating the degree of flexibility in where targets can be achieved (Table 2.2, Figure 2.6). If on the other hand vegetation types are added to the land-class feature set then a third more sites are required to meet vegetation type targets (Table 2.2). Although land-class targets can be achieved in fewer sites than vegetation type targets, i.e. is spatially more efficient at representing biodiversity, they are less effective at achieving targets for other surrogates (Table 2.4). At this scale, for all comparisons land-classes are less effective at achieving targets for other surrogates. Species, however, are the most effective surrogate in the comparison. Achieving species targets achieves most targets for all other surrogates (Table 2.4).

Comparing the total number of sites selected by a minset does not tell us much about the congruency of two features. This is because there can be several minset outcomes, so comparing the percentage of other surrogate targets achieved by a surrogate based on single minset outcomes can, at best, be regarded an estimate of what the true value would be. What is more important to consider is how well individual sites compare for the different surrogates in terms of their likelihood of being selected by a minset (i.e. irreplaceability).

Figure 2.7 indicates that both vegetation type and land-class, or their combination, perform as poor surrogates for the taxonomic biodiversity surrogates. The distribution of sites in all cases is skewed strongly in favor of the respective taxonomic surrogate. Vegetation type and land-classes perform reasonably well as surrogates for one another with about a 40% overlap of sites, but the distribution is symmetrical centered on this mean. *Conophytums* perform as excellent surrogates for endemic species with a 78% overlap of site irreplaceabilities supporting the generally held notion amongst field botanists that areas rich in *Conophytum* species are also areas rich in other range-restricted plant taxa. However, endemic taxa are overall better surrogates for *Conophytum* than *vice versa*. It is interesting to note that for a single genus with few records, *Conophytum* performed better than expected with only 15% of sites definitely required to meet endemic taxa targets (i.e. site irreplaceability =1) missed by the *Conophytum* dataset. These results mirror those for the minset outcomes (Table 2.4)

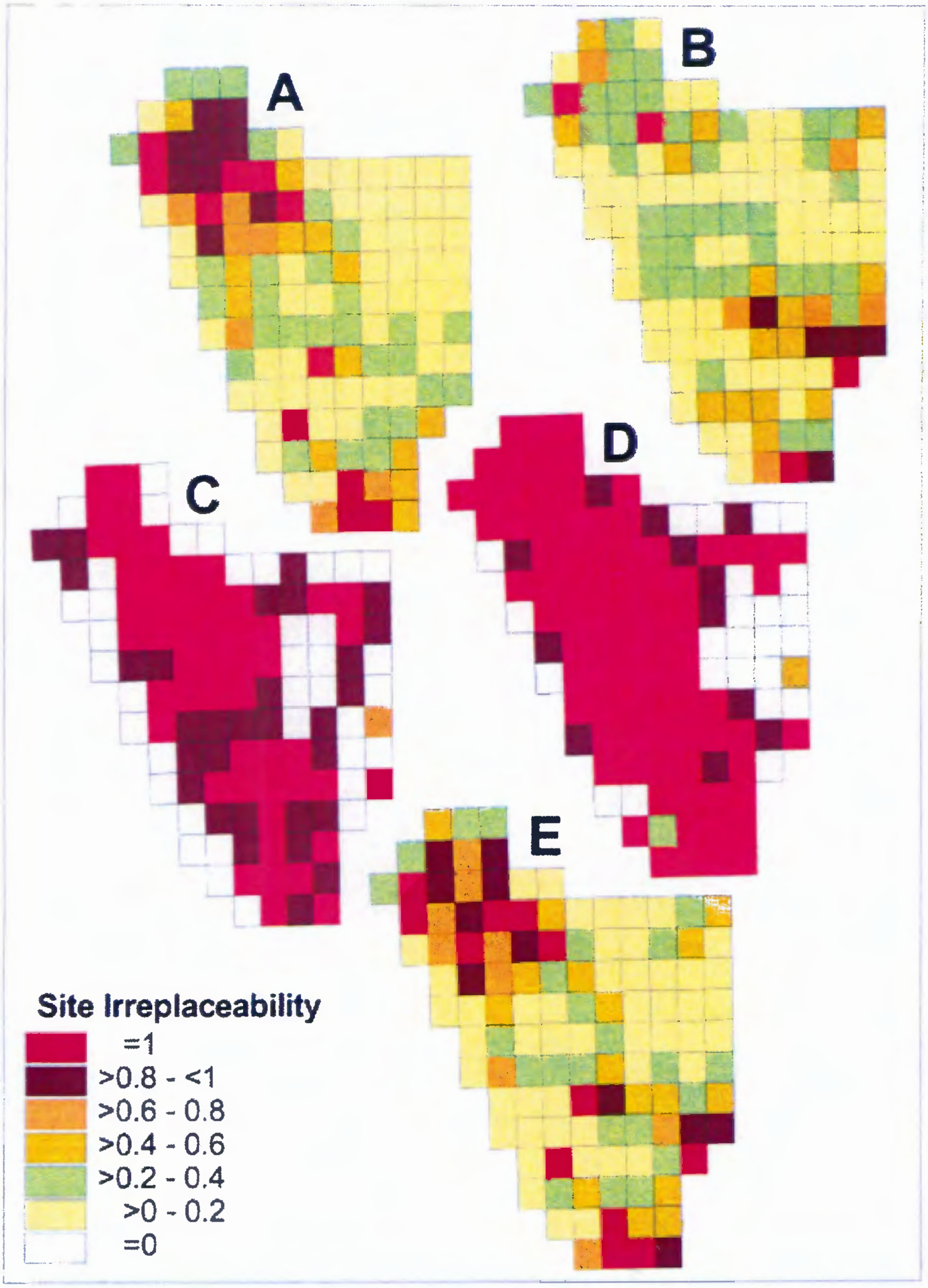


Figure 2.5: QDS-scale irreplaceability maps for (A) vegetation types; (B) land-classes; (C) *Conophytum*; (D) endemic taxa; and, (E) vegetation types and land-classes combined.

Table 2.2: The number of sites selected by a minset for the QDS dataset when using only one feature (values on the diagonal) or a combination (both row and column features targeted) of features. For example, 45 sites are required to meet VT targets whereas 47 and 78 sites are required to meet VT and LC or VT and SP targets combined, respectively. There are a total of 120 QDS sites. VT = vegetation types; LC = land-classes; C = *Conophytum*; SP = endemic species.

	VT	LC	C	SP
VT	45	-	-	-
LC	47	33	-	-
C	55	53	43	-
SP	78	79	77	75
VT+LC	-	-	58	81
VT+LC+C	-	-	-	83

Table 2.3:The number of sites selected by a minset for the SDS dataset when using only one feature (values on the diagonal) or a combination (both row and column features targeted) of features. There are a total of 1712 SDS sites. Surrogate codes are that same as for Table 2.2.

	VT	LC	C
VT	319	-	-
LC	352	340	-
C	312	337	71
VT+LC	-	-	363

Table 2.4: The percentage of row surrogate targets achieved when performing a minset to achieve the column surrogate targets. For example, in column 1, a minset that achieves 100% of VT targets also achieves 88%, 84% and 62% of the other features targets, respectively. QDS and SDS are the two spatial scales of analysis. Percentage of target achieved is calculated here as the percentage of features in each feature group that have achieved target. Surrogate codes are that same as for Table 2.2.

	QDS				SDS			
% of target achieved	VT	LC	CONO	SP	VT	LC	CONO	VT+LC
VT	100	64	79	97	100	53	14	100
LC	88	100	80	90	53	100	3	100
CONO	84	58	100	99	51	47	100	56
SP	62	43	80	100				

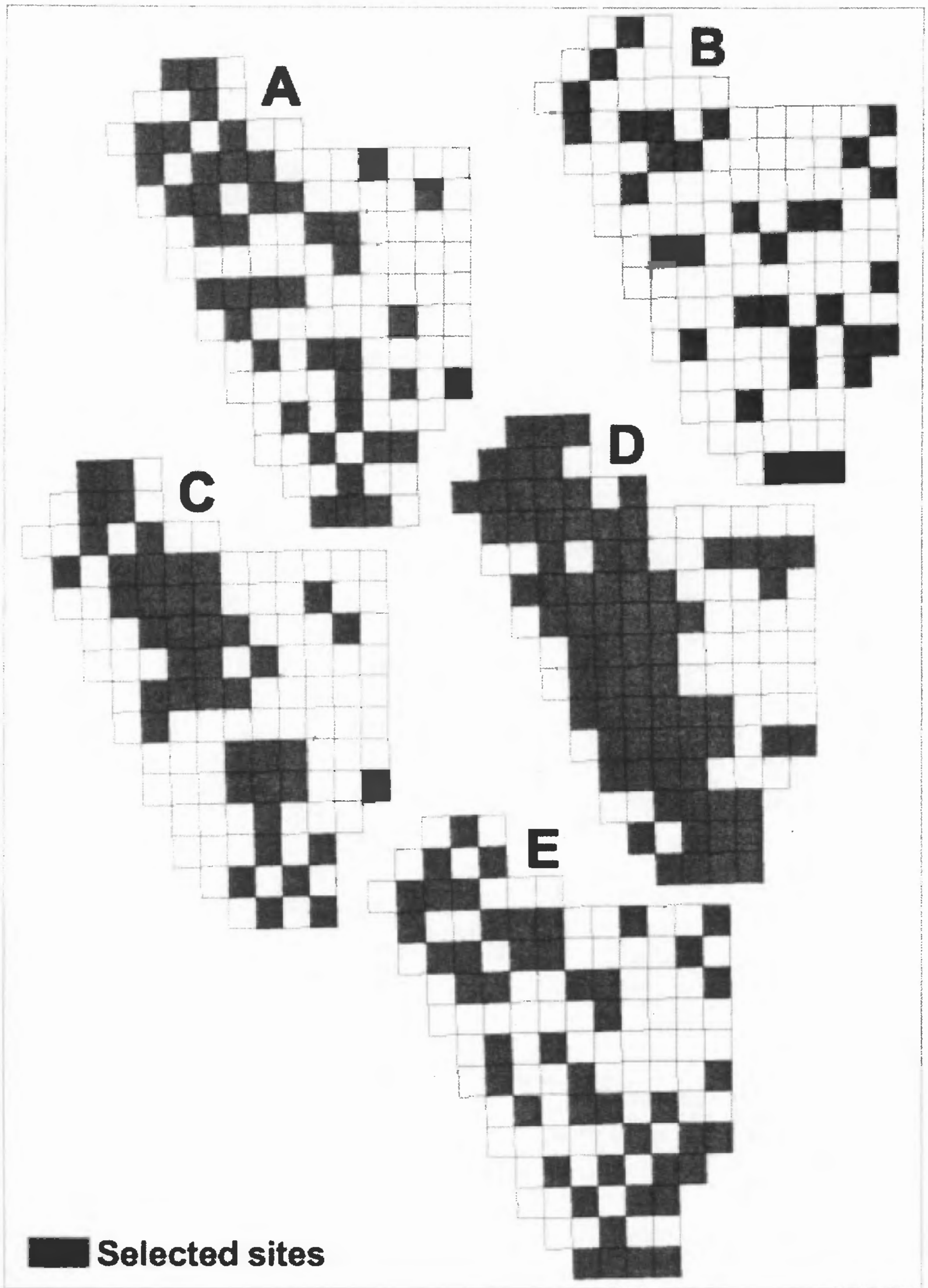


Figure 2.6: QDS-scale examples of minset outcomes for (A) vegetation types; (B) land-classes; (C) *Conophytum*; (D) endemic plants; and, (E) vegetation type and land-class combined.

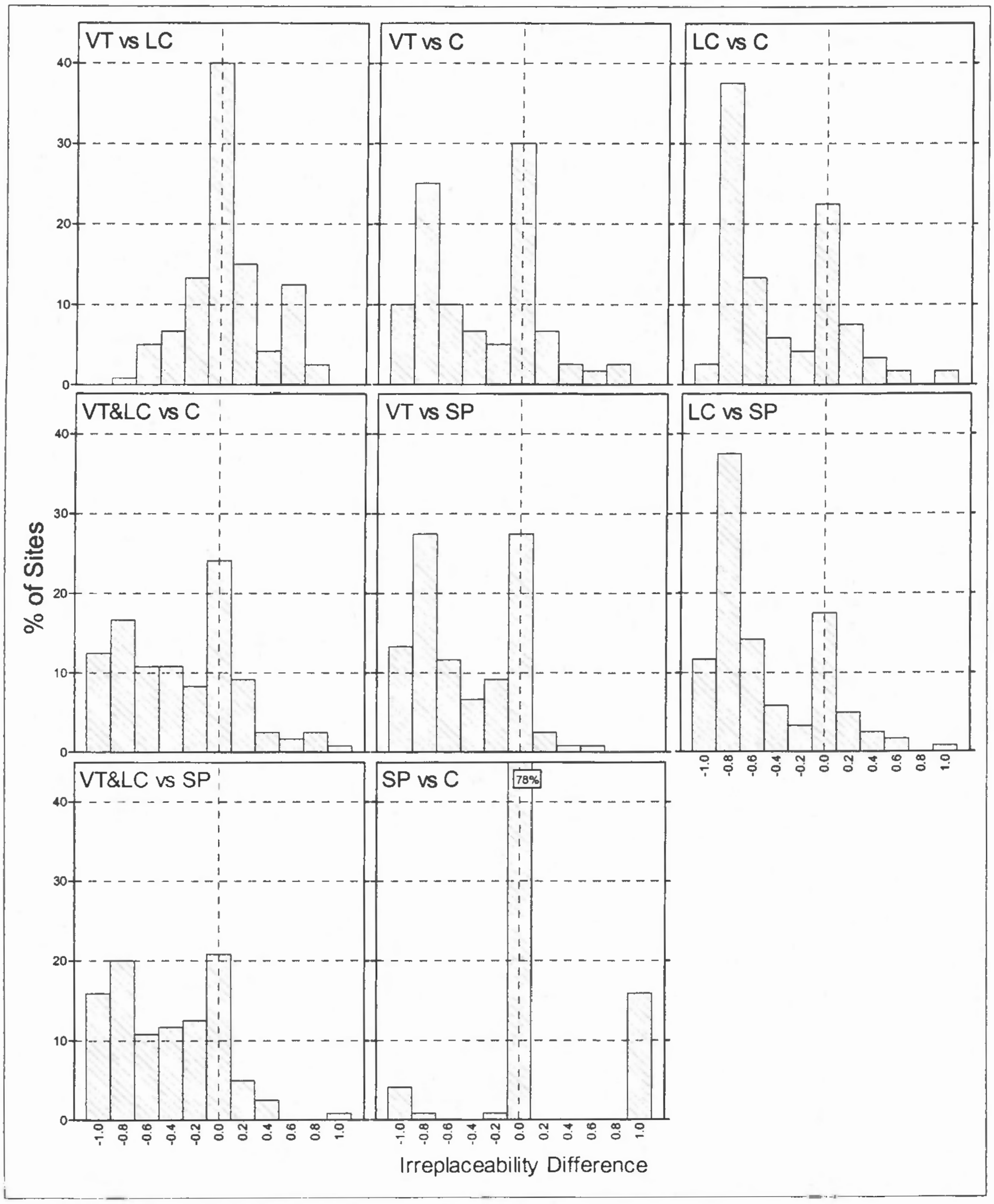


Figure 2.7: QDS-scale comparisons of the degree of congruence between two biodiversity surrogates for achieving each other's targets. The x-axis scale on each histogram is reversed when comparing how good the second surrogate is as a surrogate for the first feature (e.g. SP as a surrogate for C, or C as a surrogate for SP). The abbreviations are: VT = vegetation type; LC = land-class; C = *Conophytum*; and, SP = endemic plant taxa). Refer to Figure 2.4 for interpretation of graphs.

2.3.2 SDS-scale

At this scale the taxonomic feature performs very poorly as a surrogate for the continuous feature types. Most notably, the herbarium endemic species data is missing from the analysis. This is because data collected at the QDS scale cannot be resolved to the finer spatial scale of these analyses. The irreplaceability map for *Conophytum* (Figure 2.8) illustrates the first limitation of point taxonomic data – missing data. The large number blank sites (irreplaceability = 0, Figure 2.8) are sites that have no data. Only 21% of sites have *Conophytum* data so in any analysis based exclusively on this data the remaining 79% of sites will never be considered as options to achieve targets. This is reflected in the minsets where the targets for *Conophytum* are met within 71 sites, over four times less than the number required to meet either vegetation type or land-class targets (Figure 2.9, Table 2.3). Meeting *Conophytum* targets does not achieve many targets for the other surrogates (Table 2.4)

In contrast to the QDS scale, land-classes require more sites to achieve targets than vegetation types (Table 2.3). Why this is so is difficult to interpret. When combining these two features, the minset adds only a further 12 sites to the total to achieve both feature group's targets indicating that there is flexibility in how targets are achieved and also that there is good overlap between the two (Table 2.3). Examining the percentage targets achieved, this overlap in the absence of the other surrogate to guide where this overlap lies, only extends to about 50% of individual features for both surrogates (Table 2.4). When adding *Conophytum* to the feature set a further 11 additional sites are required to meet the targets indicating that the outstanding target can be achieved in few additional sites (Table 2.3). The combination of the two continuous feature types appears to act as a good surrogate for *Conophytum*; however, in Table 2.4 it is evident that these 11 additional sites account for 44% of the outstanding *Conophytum* target!

At this finer scale vegetation types and land-classes perform potentially as much better surrogates for each other with a 72% overlap of site irreplaceability and a symmetrical distribution of sites centered on zero (Figure 2.10). Unlike at the QDS scale, *Conophytum* is a very poor surrogate for either of the continuous surrogates with the distribution being skewed strongly in favor of both the continuous surrogates. Taxonomic data at this scale does not make a very good biodiversity surrogate. This mirrors the results of the minset outcomes in Table 2.4

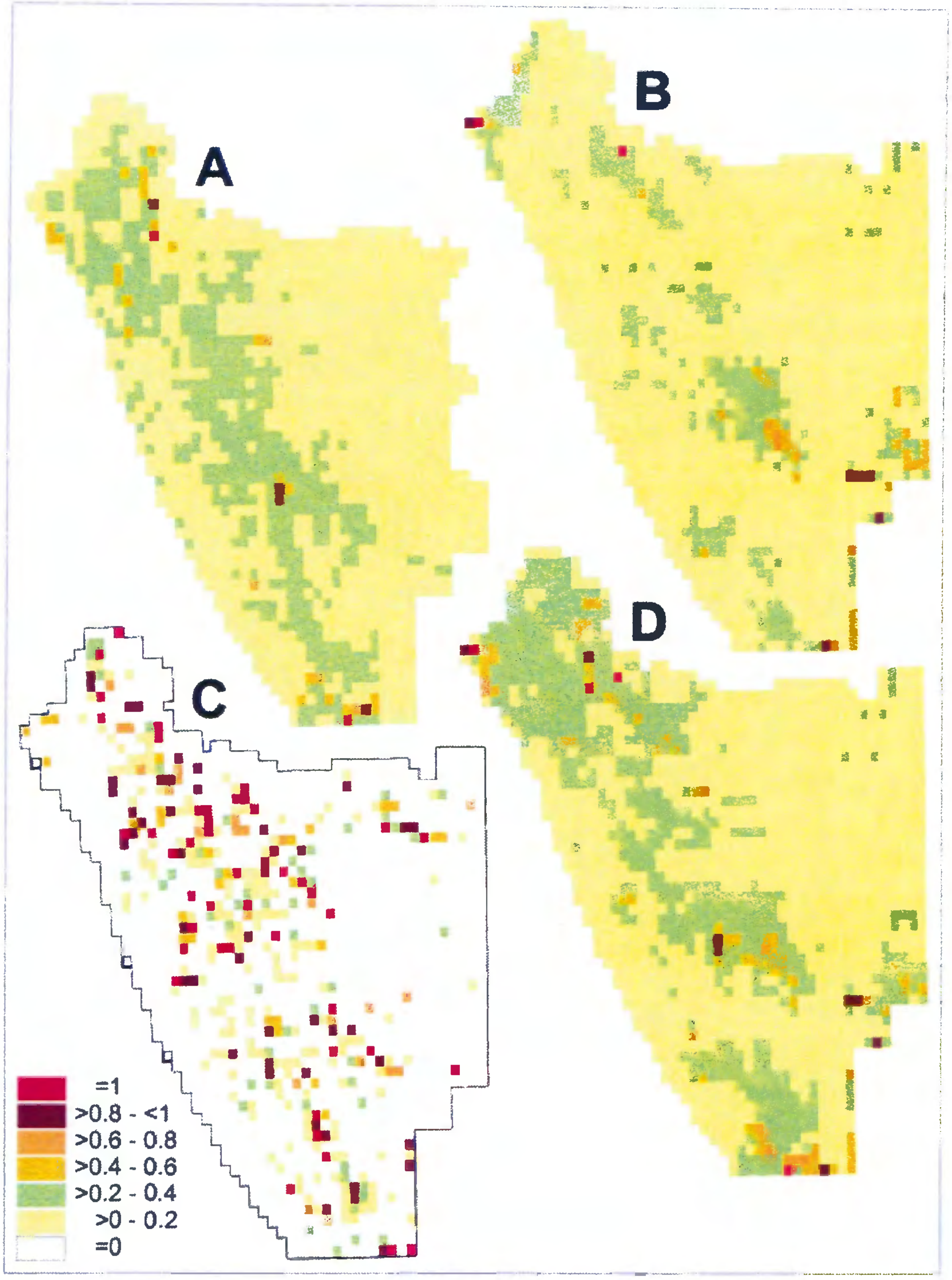


Figure 2.8: SDS-scale irreplaceability maps for (A) vegetation types; (B) land-classes; (C) *Conophytum*; and, (D) vegetation type and land-class combined.

Figure 2.9: SDS-scale examples of minset outcomes for (A) vegetation types; (B) land-classes; (C) *Conophytum*; and, (D) vegetation type and land-class combined.

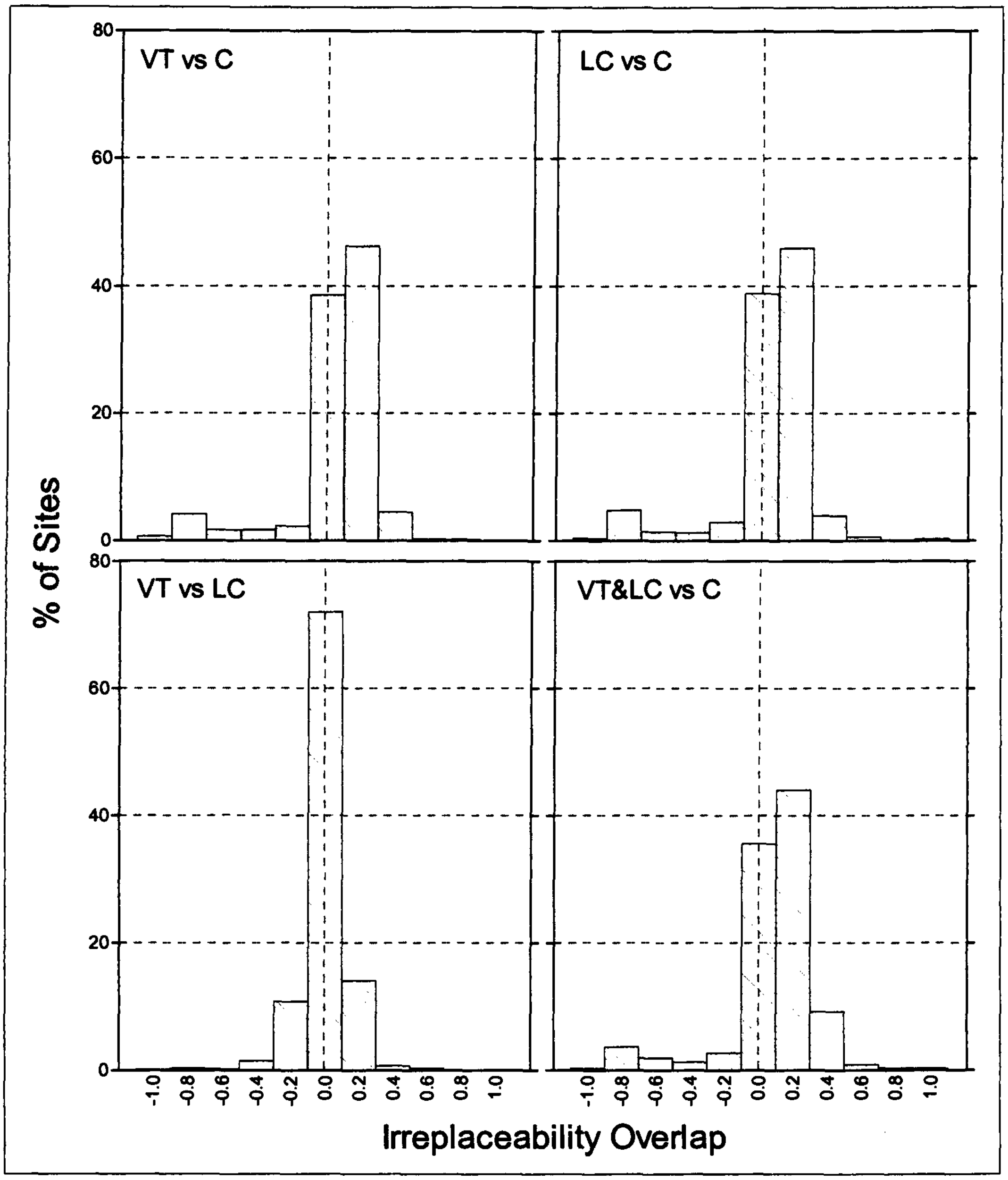


Figure 2.10: SDS-scale comparisons of the degree of congruence between two biodiversity surrogates for achieving each other's targets. The x-axis scale on each histogram is reversed when comparing how good the second surrogate is as a surrogate for the first feature (e.g. VT as a surrogate for C, or C as a surrogate for VT). The abbreviations are: VT = vegetation type; LC = land-class; and, C = *Conophytum*. Refer to Figure 2.4 for interpretation of graphs.

2.4 Discussion

The congruence of surrogates, or lack thereof, observed in this study here mirrors the general findings of the substantial literature on the subject – achieving targets for one biodiversity surrogate rarely achieves targets for all other surrogate. At the SDS scale none of the three surrogates compared performed adequately well as surrogates for each other. Achieving targets for any of the surrogates, at best, achieved 51% of targets for any other surrogate. Even combining vegetation types and land-classes only raised this value to 56% of *Conophytum* targets. Overall though, at this scale the continuous data performed better (i.e. achieved more targets for other surrogates than *vice versa*) than the taxonomic data. Understanding the distribution of *Conophytum* illustrates this congruency problem. In Namaqualand, although *Conophytum* occurs in a relatively wide range of habitats, it does not occur in any of the sandy-substrate vegetation types and by default is naturally absent from one fifth of the study area (Desmet and Hammer in prep.). It is therefore unlikely that the patterns of diversity in *Conophytum* will approximate patterns of diversity for all plant groups in the region. Lombard *et al.* (2003) also detected this problem for specific taxonomic surrogates in the fynbos.

The value of *Conophytum* as a surrogate for other biodiversity surrogates could be improved by increasing their target from 1 to say 3 occurrences. In fact, by adjusting the targets for any of the surrogates will change the outcomes of these analyses. The effect of variable targets was not examined here. It is unlikely, however, that changing targets within realistic bounds would alter the main conclusions drawn form this research.

To expect perfect congruence between surrogates is unrealistic. By their very definition surrogates, whilst being surrogates for the same entity (i.e. biodiversity), map different components of biodiversity and so lack of congruency is to be expected. It is important to look beyond this debate and understand the implications of this reality on conservation planning. No single surrogate will ever truly suffice as "the surrogate" on which to base conservation decisions. As is demonstrated here, continuous surrogates perform better than discrete point data. However, the point data despite these limitations, compliment the continuous data by highlighting areas of known biodiversity features not explicitly targeted in the continuous data.

At the QDS scale, taxonomic data make good surrogates for higher order biodiversity data in Namaqualand. This is most likely as consequence of the fact that most sites are required to achieve targets, rather than there being some meaningful underlying biogeographic explanation. However, no regional-scale conservations plans use QDS girds as planning units. These are far too coarse to make on the ground decisions. The example of using QDS taxonomic data in this study reiterates the findings of Lombard *et al.* (2003) - the sad truth about this vast pool of biodiversity data is that it is essentially useless for applied conservation decision-making. There is no biologically valid means of refining the geo-referencing of this information to make it applicable to fine-scale studies. In the age of satellite technology, GPS receivers and topographic maps it is imperative that field biologist gather herbarium and museum data at a biologically relevant scale.

At finer spatial scales continuous biodiversity information, such as the vegetation types or land-classes, are better surrogates for point species data than *vice versa.* This has important implications for the type of data used for different conservation prioritization studies. For continental or sub-continental scale broad prioritization studies it would probably be better to use taxonomic distribution data as the primary biodiversity features (e.g.Buys and Vorster 1995; Brooks *et al.* 2001; Burgess *et al.* 2002). At the fine-scale in applied studies continuous vegetation type or land-class data would be more appropriate.

For fine-scale conservation planning taxonomic biodiversity surrogates should not be used as the primary biodiversity feature on which to base decisions because they make poor surrogates in their raw form. There are a number of reasons why this type of data makes poor surrogates. Most importantly, it is impossible to sample the entire landscape. The problem of missing data and discrete locality information can be addressed by extrapolating species distributions through various modeling approaches (Cummings 2000; Bonn and Schroder 2001; Austin 2002; Ferrier *et al.* 2002b; Mac Nally *et al.* 2003). This would address both the discrete data and missing value problems. For Namaqualand and most of the world, even if species point distribution data exists it is unlikely that fine-scale modeled species distributions for conservation planning will be available for many years. This reality is demonstrated rather crudely in this study by the fact that there is only one point taxonomic dataset for Namaqualand, which is located in one of the world's major biodiversity hotspots! Also, there are problems other than time and money. Many distributions have a strong historical component (Cowling and Lombard 2002a). This makes modeling based on environmental data problematic. Furthermore, the actual determinants of pattern can be very fine scale (<10m). The environmental data most commonly used in generating models are too crude to be meaningful at this scale. Lastly, there is the problem of missing data in taxonomic datasets - does this reflect just missing data or true absences? This is an issue that can only be dealt with through well-designed dedicated field survey campaigns.

Although the continuous features are potentially better surrogates than taxonomic data at finer scales, there is the real problem of which sites to choose. If there were two sites each with 100% of their area covered by one vegetation type, but only one site was required to achieve the features target, which one would be chosen? If one of the sites contained the only known location of a *Conophytum* and each sites irreplaceability is 0.5 based on their likelihood of being needed to achieve a vegetation type target, it could be concluded that vegetation types were a good surrogate for *Conophytums* as there was a 50% chance of achieving the target for that *Conophytum* species. The congruency issue aside, the wrong site could also be selected 50% of the time.

This problem comes about because in many land-classes, species may be patchily distributed, and the reservation of part of a land-class may well miss a large number of its component species (Pressey 1994b; Ferrier *et al.* 1999; Ferrier *et al.* 2000). The patchy distribution of species can result from high levels of narrow endemism (Kirkpatrick and Brown 1994a; Lombard *et al.* 1999a), high compositional turnover along environmental and geographic gradients (Ferrier *et al*, 2000), or from historical factors (such as climatic and disturbance history) which result in relictual populations (Kirkpatrick and Brown 1994a; Araujo *et al.* 2001). This is especially true of species-rich areas such as Namaqualand (Desmet and Hammer in prep.). Therefore although land-classes provide biodiversity information for the whole landscape the decision of which sites to choose within a land-class remains problematic.

In practice both types of datasets are required for planning. Where sites are otherwise equal select the site where something in known to occur or has a high probability of occurring for modeled species distribution data rather than choosing sites randomly. It must be remembered that in the real world no two sites can ever have equal biodiversity so the more features incorporated to differentiate sites the better for planning. Solutions to the species versus land-class debate in the literature suggest a combination of the two surrogate types is better than using one or the other (Kirkpatrick and Brown 1994a;

Pressey 1994b; Lombard *et al.* 1997a; Noss 1999; Faith *et al.* 2001; Mac Nally *et al.* 2002; Williams *et al.* 2002). This may the better approach, but a trend observed in this study was that as more surrogates were added to the planning feature-set, so the number of sites required to meet targets increased. This may well be biologically valid if the goal is to conserve biodiversity, but it will have political and economic repercussions as the area required to achieve this goal gradually approaches the entire landscape.

It is interesting to note that at the fine-scale the expert derived vegetation types performed as well as the modeled land-class maps as a surrogate for one another and for the taxonomic data. This stresses the contribution that expert derived spatial data can play in the conservation planning process (Harris *et al.* 1997; Pearce 2001; Cowling *et al.* 2003d).

The novel method of comparing irreplaceability maps developed here allows for the comparison of surrogates relatively quickly using freely available software and without the need for tedious iterative computational analyses. Although the method outputs are qualitative rather than quantitative comparing surrogate combination histograms does provide an indication as to their congruency. The results and conclusions drawn using this method should, however, be checked at some point against an iterative minset procedure.

The lessons learned from this study can be summarize in a set of basic rules for incorporating biodiversity surrogates into regional conservation assessments:

Rule 1: There are no perfect biodiversity surrogates. Different surrogates focus on different aspects of biodiversity so expect different results.

Rule 2: Use as many surrogates in a conservation plan as what are available within the projects time and budget constraints.

Rule 3: As a minimum, the primary biodiversity surrogate layer should be continuous surrogate that covers the entire planning domain.

Rule 4: Do not under estimate the contribution that spatially explicit expert derived data can make to the planning process.

Rule 5: Do not discard incomplete point taxonomic datasets, as it is better to base decisions on what you know rather than what you don't know.

Rule 6: QDS scale taxonomic data is not useful for on the ground conservation decision-making.

3 Using the Species-Area Relationship to Set Baseline Targets for Conservation

3.1 *Introduction*

Targets are an integral part of contemporary conservation planning, implementation and monitoring. Systematic conservation planning is dependent on explicitness, accountability and defensibility in identifying priority conservation areas (Margules and Pressey 2000a). As a part of this, conservation targets underpin this process as they provide a clear purpose for conservation decisions, lending them accountability and defensibility (Pressey *et al.* 2003b). Targets are basically quantitative interpretations of broad conservation goals that are established in policy, by experts, implementing agencies or other stakeholders (Cowling *et al.* 1999a; Margules and Pressey 2000a; Pressey *et al.* 2003b). For example, an agency may specify that it wishes to conserve at least 10% of each vegetation type and three populations of endangered species within it's jurisdiction. Consequently, targets also provide a benchmark against which to measure the success of conservation action.

Conservation targets can be divided into two broad categories based on the scale of biodiversity surrogate targeted (Noss 1996a; Pressey *et al.* 2003a). Coarse-filter approaches set targets for features such as vegetation types, ecosystems or land-classes. Fine filter approaches use species or populations as the focal feature for conservation action (Noss 1996b). While both approaches are complimentary, for most regions limitations in species distribution datasets obligate the use of coarse filter surrogates (Lombard *et al.* 2003; Desmet *et al.* in prep.).

Vegetation or land-class maps have the advantage of covering the entire landscape, thereby eliminating the inherent spatial and taxonomic bias of species datasets (Lombard *et al.* 2003; Pressey *et al.* 2003a; Desmet *et al.* in prep.). There are limitations with using such maps. Firstly, reserve selection using the coarse filter approach is likely to protect many species for which records are deficient or are yet to be discovered. However, unless complimentary fine filter information is incorporated in the process other species, especially rarer ones, are likely to be missed (Kirkpatrick and Brown 1994b; Lombard *et al.* 2003; Desmet *et al.* in prep.). Secondly, the spatial, land-class compositional or process requirements of certain species are unlikely to be satisfied unless specifically targeted (Pressey *et al.* 2003a). Such taxa have been referred to as "focal species" (Lambeck 1997b; Boshoff and Kerley 1999; Noss *et al.* 1999a; Boshoff *et al.* 2001) or "landscape species" (Sanderson *et al.* 2002). These are essentially an alternative approach to interpreting the "umbrella" or "keystone" species concepts (Bond 1993; Mills *et al.* 1993; Launer 1994; Fleisman 2000). Thirdly, land classes do not explicitly target natural processes (Pressey *et al.* 2003b). These need to be targeted if biodiversity is to persist (Cowling *et al.* 1999a; Cowling and Pressey 2001).

These problems are compounded by problems relating to scale. Targets framed as percentages of countries or regions can be achieved while failing to protect the natural features most urgently in need of protection (Pressey *et al.* 2003b). Large regions are heterogeneous in terms of biodiversity and potential for anthropogenic transformation. Conservation areas have often been relegated to the least useable portions of regions, thereby avoiding areas where past impacts on biodiversity have been greatest and future threats are most serious (Noss *et al.* 1999b; Pressey *et al.* 2000; Scott *et al.* 2001). This is true even of regions with overall percentages under formal protection equal to or greater than 10 (Armesto *et al.* 1998; Barnard *et al.* 1998; Soule and Sanjayan 1998; Pressey *et al.* 2000; Rouget *et al.* 2003c). Also, coarse-scale maps do not capture all possible land-class combinations, so even if vulnerability over the whole area is low certain landscape biodiversity features (e.g. rare vegetation types or habitats) can fall through the cracks (Desmet *et al.* in prep.). These issues can be addressed by mapping at finer scales and with improved mapping techniques (Ferrier 2002; Desmet *et al.* in prep.). Targeting better-mapped land types with classes that are more homogeneous in terms of biodiversity and land use potential, limits the potential for conservation action to miss capturing all biodiversity (Bedward *et al.* 1992).

Mindful of these limitations, it must be accepted that for the majority of areas on this planet land-class maps of some sort will be the primary biodiversity feature used for conservation and land-use planning. So how can biologically meaningful quantitative conservation targets for land-classes be set? Whilst there are some studies dealing with a range of species (Margules *et al.* 1988; Saetersdal *et al.* 1993; Travaini *et al.* 1997), minimum viable population (Nunney 1993; Boshoff *et al.* 2001; Burgman *et al.* 2001), meta population (Lindenmayer and Lacy 1995), genetic diversity (Lacy 1997; Ferguson *et al.* 1998), community (Prins *et al.* 1998), habitats (Turner *et al.* 1999; Calkin *et al.* 2002) or ecosystem (Turner *et al.* 1992; Noss 1996b) targets, there is generally a paucity of work dealing specifically with targets for land-classes.

The widely used 10% target, recommended by IUCN, when applied to land-classes implies that all are equal in terms of their species diversity, abundance and distribution. This is certainly not the case. More questionable though, is the biological foundation for this target. Despite the potential arbitrariness of this target (Soule and Sanjayan 1998; Pressey *et al.* 2003b), the origin is partly founded in the original work on the species-area relationship. A general "rule of thumb" first noted by Darlington (1957) and developed from early observations of species-area relationships was that a ten-fold decrease in area resulted in a two-fold decrease in species, or alternatively 10% of area would conserve 50% of species (Diamond and May 1976). Even if at some point this observation influenced the conception of the IUCN 10% target at the World Parks Congress in Caracas in 1993, the question arises as to whether saving 50% of the planet's terrestrial species is really adequate? General consensus in the literature is that 10% of area is not sufficient to represent the majority of biodiversity assuming that the remainder of the landscape is cleared or not conserved (Soule and Sanjayan 1998; Rodrigues and Gaston 2001).

Is it possible to set ecologically meaningful targets for land-classes? In an attempt to address this question this paper returns to one of ecology's oldest observations, the species-area relationship (SAR), to find answers.

"You will find more species if you sample a larger area" (Rosenzweig 1995). This could also be stated as follows: "you will conserve more species if you conserve a larger area". Patterns in the SAR are well explored in the ecological literature across a range of spatial and temporal scales (Rosenzweig 1995; Lomolino 2000a; Knowles 2001; Lomolino 2001c; Collins *et al.* 2002; Haila 2002; Lomolino 2002). Attempts have been made to develop a functional understanding of the SAR (e.g. Harte 1999). Although, there is as yet no widely accepted ecological theory to explain the relationship, the basic pattern is real. Many mathematical models can and have been used to describe the SAR (Lomolino 2000b), however, the power relationship or Arrhenius equation is probably the most popular in the literature (Rosenzweig 1995):

Equation 1

$$\text{Species} = \text{k.Area}^{z}$$

In this relationship, k is a scaling factor that relates to sample size used (Rosenzweig 1995; Lomolino 2000a). The meaning of k is debatable (Gould 1979; Lomolino 2000a; Lomolino 2001a), however, I will not discuss this here. The rate at which species are encountered in a system is described by the parameter z (Rosenzweig 1995; Lomolino 2000a). The SAR has been applied to a host of questions in ecology. Those with a direct conservation angle range from the "Single Large Or Several Small" debate (Diamond and May 1976; Bond *et al.* 1988; Deshaye and Morisse 1989; Cowling and Bond 1991) to predicting the loss of species from fragmented landscapes (Tilman *et al.* 1994; Kemper *et al.* 1999; Cornelius *et al.* 2000; Miller and Cale 2000; Magura *et al.* 2001; Acosta and Robertson 2002; Haila 2002; Tscharntke *et al.* 2002; Zurlini *et al.* 2002).

Fragmentation research has come very close to actually addressing the target issue by using Equation 1 to estimate the proportion of species lost given a reduction in area (Brooks *et al.* 1999; Kinzig and Harte 2000). From the conservation planning perspective, rather than asking how many species will be lost if a landscape is fragmented, the question can be turned around to ask how many species will be gained for protection if some more landscape is added to the conservation network?

Diamond and May (1976) used the power equation to predict the number of species remaining if a given percentage of a landscape was transformed. If the z-value for a biota or land-class is known then by using Equation 2 it is possible to predict the proportion of species remaining if the area is reduced by a given proportion.

Equation 2:

$$S' = A'^{z}$$

Here S' and A' denote the proportion of species and area rather than absolute values. This equation can be reordered to address conservation targets to determine the proportion of area required to represent a given percentage of species:

Equation 3:

$$A' = \sqrt[z]{S'} \quad \text{or} \quad \text{Log}A' = \text{Log}S'/z$$

For the SAR rule of thumb discussed above - a ten fold decrease in area equates to a two fold decrease in species - the z-value is approximately 0.3. Using this same z-value but changing the proportion of species targeted to 75% it follows from Equation 3 that the proportion of area required increases from 10% to 38%. Increasing the species target to 95% then the area target becomes 84%! These are quite significant changes in area required to meet this basic biodiversity target of representing each species at least once. Published z-values for biotas range between approximately 0.1 and 0.4 (Rosenzweig 1995). Although this range in the exponent is small, the nature of the power equation means that for a species target of 75%, the area target can range from 5% to 48%, respectively.

It is possible to use the SAR to set conservation targets for land-classes. The method for setting this target involves estimating the area of a land-class that is required to represent a given proportion of the species occurring in the land-class. Thus, it remains a question of being able to calculate the z-value of the SAR for a land-class. To achieve this it is necessary to generate a SAR curve based on some form of inventory data. As the curve of the power model is a straight-line in log-log space and the slope is the z-value, it is possible to calculate the z-value without the need to generate the actual curve using species-area data that samples larger and larger proportions of a land-class. There is no need to demonstrate the relationship by fitting the power curve as the suitability of the model is assumed a priori. For the log transformation of the power model the slope of the curve, hence the z-value, can be calculated from the formula:

Equation 4

$$z = (y_2 - y_1)/(x_2 - x_1)$$

where y_2 = log(total number of species in a land-class); y_1 = log(average number of species per sample); x_2 = log(total area of land-class); and, x_1 = log(average area of samples). Using inventory data three of these variables are known and all that remains is to estimate the total number of species that occur in the vegetation type. Several nonparametric estimator functions are available for estimating the true number of species in an area based on a set of random samples (Colwell and Coddington 1995; Colwell 1997).

In this paper phytosociological survey data are used to calculate the z-values for land classes (Succulent Karoo biome vegetation types) using Equation 1 and by estimating the true number of species per vegetation type using EstimateS software (Colwell 1997). As a first step, the accuracy of these estimates is explored using model datasets with species-abundance distributions similar to those observed in the survey data. Secondly, z-values for vegetation types with sufficient survey sites to allow for meaningful estimates to be made are estimated. Following from this, observed z-values are extrapolated to other vegetation types in the Succulent Karoo that do not have survey data. This is achieved by relating z-values to landscape physical properties that act as surrogates for geographic species turnover and habitat diversity. Finally, some generalisations about land-class characteristics are made that can be applied to other conservation situations where suitable survey data is lacking.

3.2 Methods

3.2.1 Estimating true species number

To determine how well various non-parametric functions estimate true species richness, estimates using random samples of five model datasets were compared. Each dataset had the same total number of species, but differed in the species-abundance distribution and location of species in the landscape. The advantage of using these model systems to explore estimates is that the total number of species, their location and relative abundance are known. Five model "land-classes" were constructed. Three had species randomly located in the landscape (1 patch), one had one gradient (2 patches) and the other two gradients (4 patches) of species turnover. The species abundance function approximated a log-normal distribution. One of the random and both the gradient models had the same abundance distribution. The remaining two random models had proportionately more rare species, thus shifting the mode of the distribution to the right. In all models the total number of species was 100 and individuals 40 000. A sampling grid of 2500 cells was overlain on each model land-class and a summary site by species database was generated. From this database, samples of 250, 50, 25, and 15 "plots" were randomly selected for analysis. These samples correspond to 10, 2, 1 and 0.6% of total area, respectively.

The software package EstimateS (Colwell 1997) was used to generate estimates of the total number of species present in the model dataset based on the samples of survey plots. EstimateS provides the user with ten non-parametric statistical functions to estimate the true number of species based on species incidence in a sample of survey sites. These estimators are discussed in Colwell and Coddington (1995). Only seven estimators (ACE, ICE, Chao1, Chao2, Jack-knife1, Jack-knife2 and Bootstrap) were used in the comparative analyses, as the others (Michaelis-Menten runs, Michaelis-Menten means and Cole) have not been widely tested for this purpose in the literature.

To determine which estimator to use with the real-world data estimators were compared by plotting the difference between actual and estimated true species number against different combinations of the five variables produced by EstimateS that characterise the sample data. These variables are: (1) number of samples; (2) observed number of species in sample dataset; (3) observed number of individuals in sample dataset; (4) number of singletons (i.e. species that only occur once in the sample dataset); and, (5) number of doubletons (i.e. species that only occur twice in the sample dataset. Furthermore, for each dataset, the difference between the estimated true species number and the actual species number (100 species for all datasets) were related to the ratio between the proportion of doubletons and the total observed species described by the following formula:

Equation 5:

(doubletons/(singletons+doubletons))/total observed species

Of all possible combinations of the five descriptive dataset variables provided by the EstimateS, this ratio produced the most significant relationship between estimation error and sample data properties.

3.2.2 Estimating z-values for selected Succulent Karoo vegetation types

Phytosociological relevé data from the Succulent Karoo biome of South Africa were used to estimate true species number for Succulent Karoo vegetation types. The relevé database contained 5491 georeferenced survey sites from 25 different studies conducted in the biome over the last 20 years. The Succulent Karoo biome of southern Africa was expertly classified and mapped into 132 vegetation types (Driver *et al.* 2003b). Each relevé was assigned to a vegetation type based on its geographic location. In total, 42 vegetation types each with more than 30 relevés were used for the analyses. The 30-sample limit was determined in exploratory analyses as being the smallest average number of samples where the standard deviation of the final estimate was less than 5% of the estimate.

The estimated true species number was used in Equation 4 to calculate the z-value of the SAR for each vegetation type.

3.2.3 Extrapolating z-values to all Succulent Karoo vegetation types

Ultimately, the goal of this exercise is to generate conservation targets for all land-classes within a biome or planning domain. The problem faced in the Succulent Karoo as well as most of the rest of the world is that there are inadequate survey data for all land-classes. For this Succulent Karoo example, 42 out of 132 vegetation types had 30 or more survey sites with only nine having more than 100 relevés. Therefore it is necessary to extrapolate z-values to vegetation types within a biogeographic province based on an observed relationship between the z-values for vegetation types or land-classes and some remotely measurable land-class properties that could explain patterns of diversity.

The effect of area has already been accounted for in the z-value. Species diversity also relates strongly to habitat diversity (Rosenzweig 1995). Therefore, it makes sense to introduce a variable that can act as a surrogate for habitat diversity. Z-values are related the two independent measures of land-class habitat diversity. The first is a simplistic measure of topographic diversity summarised in two ways: (a) the standard deviation of the mean altitude for a vegetation type as determined from a 100m grid cell resolution digital elevation model; and, (b) the ratio of volumetric to planimetric surface area per vegetation type determined from the same elevation model. The hypothesis is that more topographically heterogeneous land-classes would have more habitats and, consequently, would support more species per unit area and have higher the z-values.

The second measure of habitat diversity is a count of the number of ecological land-classes present within each vegetation type. The ecological zones were developed using a generalised dissimilarity modelling technique discussed in Ferrier (2002) and applied by Ferrier *et al.* (in prep.) for the Namaqualand region of the Succulent Karoo. Each zone is determined as a function of remotely determined topographic, edaphic and climatic variables scaled according to observed patterns of plant diversity. Each zone can be considered environmentally and biologically homogeneous relative to other zones, thus a count of the number of zones represented in a vegetation type can be regarded as an alternative, albeit crude measure of habitat diversity.

In addition to the three habitat or beta diversity variables, latitude and longitude were introduced as geographic or gamma diversity variables. For the analyses the geographic centroid of each vegetation type was used. These variables are useful at the landscape scale, as patterns of gamma diversity relate strongly to distance between areas (Ferrier *et al.* 2002a)

To develop the relationships between z-values and the landscape variables, the 42 vegetation types were grouped into eight higher order vegetation type categories (Table 3.1). The groups were based on known biogeographic and physiognomic similarities in vegetation types. Regression models relating z-values to the environmental variables were built using linear, non-linear and generalised additive modelling methods. S-Plus statistical package (http://www.mathsoft.com/splus/) was used to perform analyses.

Table 3.1: The classification of vegetation types into higher order vegetation categories. Vegetation types in each category were used to develop a regression model relating z-values for the category to remotely determined landscape variables.

Vegetation Type	Vegetation Category
Bushmanland Arid Grassland Karas Upland Nama Karoo	Bushmanland Nama Karoo
Eastern Little Karoo Vanwyksdorp Gwarrieveld Western Spekboomveld	Gwarrieveld
Kamiesberg Mountain Brokenveld	Kamiesberg Brokenveld
Anysberg Quartz Patches Langeberg Quartz Patches Warmwaterberg Quartz Patches	Little Karoo Quartz-patches
Prince Albert Succulent Karoo Southern Tanqua Karoo Western Little Karoo	Little Karoo Succulent Karoo
Alexander Bay Gravel Patches Eastern Bushmanland Quartz and Gravel Patches Knersvlakte Quartzfields Lekkersing Quartz Patches Riethuis Quartzfields	Namaqualand Quartz-patches
Central Knersvlakte Lowland Succulent Karoo Central Richtersveld Succulent Karoo Knersvlakte Shales Namaqualand Klipkoppe Namaqualand Klipkoppe Flats Namaqualand Lowland Succulent Karoo Northern Knersvlakte Lowland Succulent Karoo Northern Richtersveld Lowland Succulent Karoo Nuwerus Quartzite Succulent Karoo Richtersberg Mountain Desert Rooiberg Quartzite Succulent Karoo Southeastern Richtersveld Quartzites Southern Knersvlakte Lowland Succulent Karoo Southern Richtersveld Lowland Succulent Karoo Tanqua Karoo Upper Annisvlakte Succulent Karoo West Gariep Desert West Gariep Lowlands	Namaqualand Succulent Karoo
Lamberts Bay Strandveld Namaqualand Coastal Dunes Namaqualand Red Sand Plains Namaqualand Southern Strandveld Northern Richtersveld Yellow Dunes Richtersveld Red Dunes Richtersveld White Dunes	Sandveld

3.3 Results

3.3.1 Determining the best estimator of true species number

Errors in the estimation of true species number are dependent on data properties (Figure 3.1). This error is broadly consistent between all estimators, except the Bootstrap estimator; however, species distribution patterns (random vs. patchy) result in two distinct error patterns. For the random species distributions species with some or many rares in the original dataset, the ratio is consistently high, and the estimators under estimate true species number as the ratio (viz. the proportion of rare species in the sample dataset) increases. The number of rares in the sample dataset (i.e. singletons and doubletons) increases as the proportion of original data sample decreases (i.e. as sample size decreases). Conversely, in the random and patchy species distributions with few rare species the estimators over estimate the true species number as the ratio increases. No relationship using combinations of the five descriptive variables could be found that clearly differentiates these two responses. Only the Bootstrap estimator shows a unitary response across all datasets. It is important to note that below a ratio of approximately 0.007 all estimators show a similar pattern in error that is within a 10% under estimation, or over estimate of 10% for Jack-knife 1, of the true species number. For all estimators a regression using all datasets with ratios below 0.007, the slope of the regression is not significantly different from zero.

For the purposes of estimating the true species number for the Succulent Karoo data the Bootstrap estimator was used because, for all model datasets, this estimator shows a unitary error response to sample properties.

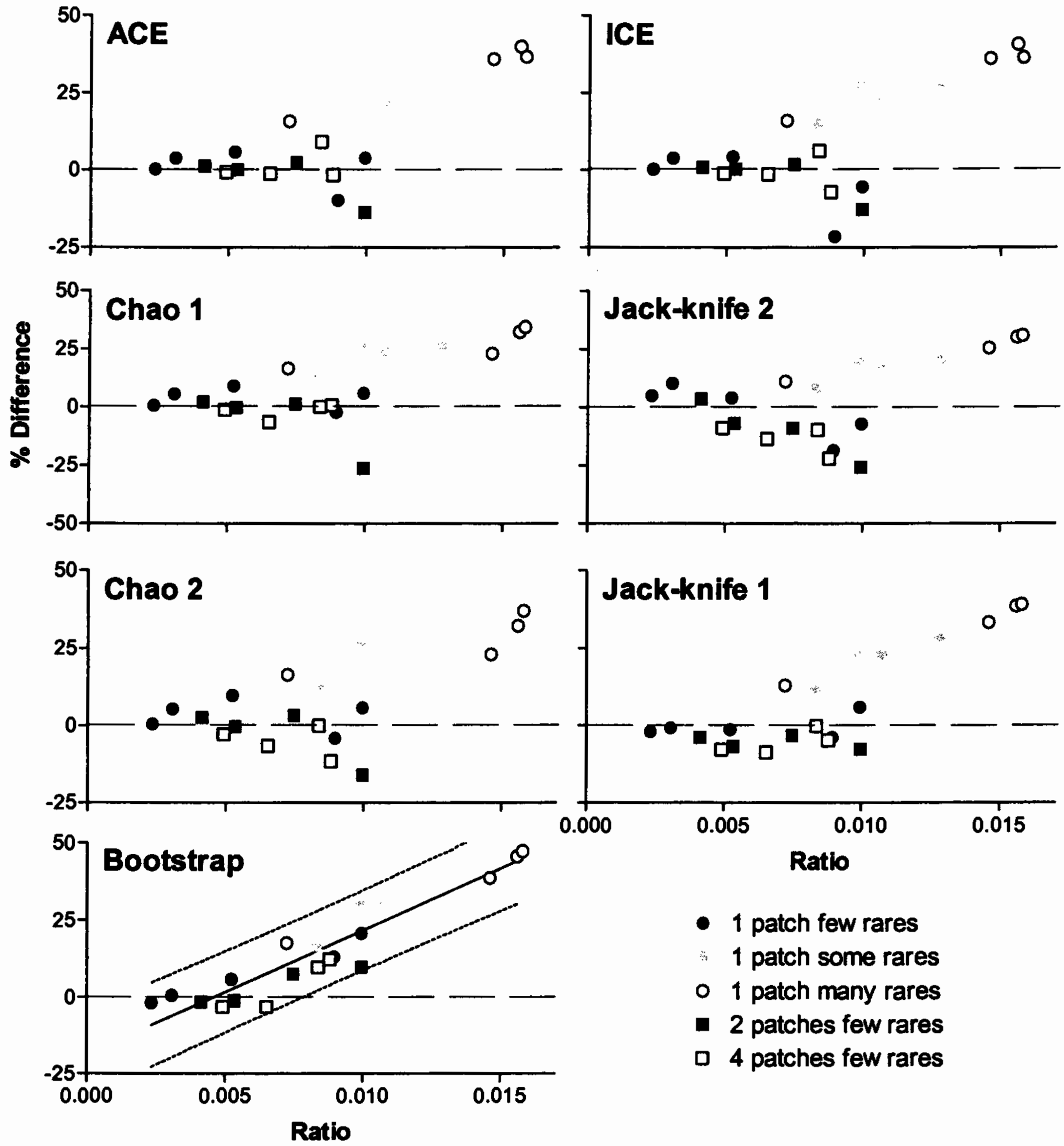


Figure 3.1: The relationship between the estimator error (i.e. percentage difference between true and estimated species number) for seven estimators, and properties of the sample datasets for the different sized sub-samples of the five model land-classes. The ratio is the proportion of doubletons relative to the total number of observed species as described by Equation 5.

3.3.2 Estimating true species number for Succulent Karoo vegetation types data

Estimates of true species number for Succulent Karoo vegetation types are presented in Table 3.2. For all vegetation types the error estimation ratio is well below the 0.007 inflection point observed in the model datasets. Based on the model analyses this would suggest that the estimates of true species number are at least within 10% of actual values. However, the ranges in estimates for the different estimators are quite large and in some cases up to 50% of the mean estimate (Table 3.2). This would suggest that the ratio does not apply to real-world data where the data properties are different.

The Bootstrap estimator is consistently the lowest of the seven estimates of true species number. This estimate is for the most part within or close to the lower standard deviation limit of the other estimates. The Bootstrap estimate has to be regarded as the most conservative estimate of true species number. This will have impacts on the interpretation of the resultant targets.

The z-value was calculated for each vegetation type using Equation 4. The average relevé size was taken to be 10x10m or $100m^2$. Varying the relevé area by an order of magnitude (i.e. 10 to $1000m^2$) changed the resultant z-value by less than 0.01%.

Table 3.2: Estimates of true species number for 42 Succulent Karoo vegetation types. Values are rounded to the nearest whole number. For explanation of the data property variables see text. Only mean values for ACE, ICE, Jack-knife 2 and Bootstrap are calculated by EstimateS. The z-value was calculated using the Bootstrap estimator.

Species Data	Data Properties								Estimates										Z	Estimate Range		
Succulent Karoo Vegetation Types	No. of relevés	Avg. No. of species/plot	STD	Total no. of observed species	Individuals	Singletons	Doubletons	RATIO	ACE	ICE	Avg. Chao1	SD	Avg. Chao2	SD	Avg. Jack1	SD	Jack2	Bootstrap	z-value estimate	Estimate range	Mean estimate	Range as % of mean
Alexander Bay Gravel Patches	73	12	5	145	855	54	18	0.00172	209	210	220	31	220	31	198	11	234	168	0.18	66	208	32
Anysberg Quartz Patches	153	21	13	218	2765	44	23	0.00157	249	249	257	17	257	17	262	8	283	239	0.166	44	257	17
Bushmanland Arid Grassland	59	6	5	151	443	78	27	0.0017	287	290	258	35	258	35	228	24	277	184	0.184	106	255	42
Central Knersvlakte Lowland Succulent Karoo	35	7	5	84	252	42	8	0.0019	137	139	180	53	180	53	125	12	157	101	0.183	79	146	54
Central Richtersveld Succulent Karoo	163	12	9	394	1911	170	56	0.00063	624	626	646	55	646	55	563	27	676	466	0.228	210	607	35
Eastern Bushmanland Quartz And Gravel Patches	66	50	44	320	2802	63	30	0.00101	363	365	383	22	383	22	382	24	414	349	0.162	65	377	17
Eastern Little Karoo	30	15	3	58	466	14	10	0.00718	68	68	66	7	66	7	72	4	76	65	0.099	11	69	16
Kamiesberg Mountain Brokenveld	49	21	9	313	995	138	66	0.00103	492	497	454	33	454	33	448	25	519	374	0.171	145	463	31
Karas Upland Nama Karoo	31	9	5	80	308	27	16	0.00465	104	105	101	11	101	11	106	7	117	92	0.137	25	104	24
Knersvlakte Quartzfields	477	10	8	311	4101	84	45	0.00112	393	394	387	23	387	23	395	13	434	350	0.221	84	391	21
Knersvlakte Shales	61	7	3	130	434	55	29	0.00266	199	201	180	19	180	19	184	12	210	155	0.197	55	187	29
Lamberts Bay Strandveld	312	23	6	175	7603	76	16	0.00099	364	364	343	63	343	63	251	46	310	205	0.145	159	311	51
Langeberg Quartz Patches	68	10	8	182	735	56	25	0.0017	224	226	241	22	241	22	237	10	268	207	0.209	61	235	26
Lekkersing Quartz Patches	68	9	4	114	582	33	28	0.00403	142	142	132	9	132	9	147	10	152	130	0.172	22	140	16
Namaqualand Coastal Dunes	62	15	6	197	925	68	36	0.00176	264	266	259	21	259	21	264	17	295	228	0.172	67	262	26
Namaqualand Klipkoppe	282	13	11	640	3801	251	104	0.00046	939	940	939	51	939	51	890	35	1036	750	0.221	286	919	31
Namaqualand Klipkoppe Flats	150	8	6	244	1247	106	35	0.00102	384	385	399	43	399	43	349	33	420	289	0.208	131	375	35

Species Data	Data Properties								Estimates											Z	Estimate Range		
Succulent Karoo Vegetation Types	No. of relevés	Avg. No. of species/plot	STD	Total no. of observed species	Individuals	Singletons	Doubletons	RATIO	ACE	ICE	Avg. Chao1	SD	Avg. Chao2	SD	Avg. Jack1	SD	Jack2	Bootstrap	z-value estimate	Estimate range	Mean estimate	Range as % of mean
Namaqualand Lowland Succulent Karoo	53	16	14	358	932	157	78	0.00093	524	529	513	33	513	33	512	24	590	428	0.196	162	516	31
Namaqualand Red Sand Plains	151	24	16	603	3164	249	88	0.00043	923	926	950	61	950	61	850	48	1010	709	0.195	301	903	33
Namaqualand Southern Strandveld	48	15	11	186	712	66	39	0.002	254	256	240	18	240	18	251	14	277	216	0.195	61	248	25
Northern Knersvlakte Lowland Succulent Karoo	62	6	3	94	424	36	14	0.00298	135	136	136	21	136	21	129	9	151	110	0.174	41	133	31
Northern Richtersveld Lowland Succulent Karoo	70	10	6	147	586	75	21	0.00149	268	270	273	44	273	44	221	20	274	178	0.198	96	251	38
Northern Richtersveld Yellow Dunes	130	11	6	138	1450	51	15	0.00165	194	195	218	34	218	34	189	12	224	160	0.17	64	200	32
Nuwerus Quartzite Succulent Karoo	35	42	32	287	1444	61	41	0.0014	323	325	331	15	331	15	346	15	366	316	0.182	50	334	15
Prince Albert Succulent Karoo	137	12	4	94	1726	8	8	0.00532	97	97	97	4	97	4	102	3	102	98	0.123	5	99	5
Richtersberg Mountain Desert	75	8	5	207	646	103	31	0.00112	358	361	371	47	371	47	309	22	379	250	0.221	129	343	38
Richtersveld Red Dunes	98	7	2	90	706	44	17	0.0031	161	161	143	23	143	23	134	14	160	109	0.186	52	144	36
Richtersveld White Dunes	51	15	7	122	815	30	16	0.00285	144	145	148	14	148	14	151	6	165	136	0.157	29	148	20
Riethuis Quartzfields	51	15	11	205	615	92	45	0.0016	330	334	296	26	296	26	295	23	341	245	0.193	96	305	31
Rooiberg Quartzite Succulent Karoo	41	10	6	109	380	38	20	0.00316	142	143	143	15	143	15	146	8	164	126	0.178	38	144	26
Southeastern Richtersveld Quartzites	152	7	4	235	1244	85	27	0.00103	325	325	363	40	363	40	319	19	377	271	0.237	106	335	32
Southern Knersvlakte Lowland Succulent Karoo	55	8	3	98	432	32	16	0.0034	124	125	127	15	127	15	129	7	145	113	0.164	32	127	25
Southern Richtersveld Lowland Succulent Karoo	41	6	7	116	298	46	31	0.00347	164	165	148	13	148	13	161	13	176	138	0.196	38	157	24
Southern Tanqua Karoo	120	17	6	133	1849	31	21	0.00304	162	162	154	11	154	11	164	6	174	148	0.134	26	160	16
Tanqua Karoo	272	11	6	299	3101	89	44	0.00111	382	383	386	25	386	25	388	14	433	339	0.19	94	385	24
Upper Annisvlakte Succulent Karoo	36	10	5	114	348	60	19	0.00211	216	220	203	34	203	34	172	17	212	139	0.182	81	195	42
Vanwyksdorp Gwarrieveld	47	31	10	143	1389	38	14	0.00188	175	176	190	23	190	23	180	12	203	159	0.103	44	182	24
Warmwaterberg Quartz Patches	122	8	7	177	1254	46	21	0.00177	212	213	224	20	224	20	223	9	247	198	0.208	49	220	22
West Gariep Desert	313	9	6	295	3099	64	46	0.00142	341	341	338	15	338	15	359	12	377	326	0.217	51	346	15
West Gariep Lowlands	63	9	6	121	499	51	19	0.00224	186	187	185	26	185	26	171	13	202	143	0.181	59	180	33

Species Data	Data Properties								Estimates										Z	Estimate Range		
Succulent Karoo Vegetation Types	No. of relevés	Avg. No. of species/plot	STD	Total no. of observed species	Individuals	Singletons	Doubletons	RATIO	ACE	ICE	Chao1 Avg.	Chao1 SD	Chao2 Avg.	Chao2 SD	Jack1 Avg.	Jack1 SD	Jack2	Bootstrap	z-value estimate	Estimate range	Mean estimate	Range as % of mean
Western Little Karoo	162	25	11	291	4111	66	40	0.0013	342	343	343	18	343	18	357	12	383	322	0.148	61	348	18
Western Spekboomveld	35	29	9	142	1019	35	12	0.0018	168	170	188	24	188	24	176	12	198	157	0.101	41	178	23

3.3.3 Extrapolating results to the bio-region

In order to extrapolate the observed patterns in z-values to all vegetation types in the Succulent Karoo, the z-values were related to the five landscape environmental variables (Figure 3.2). In Figure 3.2 the z-values are classified into higher order vegetation categories that reflect biogeographic and physiognomic similarities in the vegetation (Table 3.1). There do appear to be relationships between observed z-values per vegetation type category and the landscape variables examined. Regression models were built for the Namaqualand Succulent Karoo; Sandveld; and, Namaqualand quartz patches vegetation categories. These were the only models that were significant given the number of data points (i.e. vegetation types) available fore each regression. For all three cases the basic model used to extrapolate values expresses the z-value, firstly, as a function of geographic location and, secondly, as a function of topographic diversity.

For the Namaqualand Succulent Karoo vegetation types, a generalised additive model was used to build the relationship between z-values and the landscape variables (Figure 3.3, **Error! Reference source not found.** 1). Both ALT (the standard deviation of mean vegetation type altitude) and RATIO (the ratio between volumetric and planimetric surface area of each vegetation type) proved to be almost equally significant. ALT was the variable eventually used as the spread of points along the x-axis is more even than for RATIO. The significance of the model using ALT was also marginally more significant than the model with RATIO ($p = 7.32e-009$ vs. $p = 1.496e-008$). The non-zero slope in the residuals would indicate that is still a fourth significant variable missing from the model explaining the pattern in z-values.

For Namaqualand Quartz patch vegetation types (Figure 3.4, Appendix 3.1), a linear model using only longitude proved to be reasonably effective at capturing the variation in z-values. The Eastern Bushmanland Quartz and Gravel Patches vegetation type was included in this vegetation category. However, if this vegetation type is excluded there is an improvement in the fit the model despite the loss of one degree of freedom (Figure 3.5; Appendix 3.1). The Bushmanland quartz patches should probably be considered a separate biogeographic region much like the Little Karoo quartz patches are (Schmiedel 2002). Neither of the topographic diversity variables demonstrated any significant relationship with z-values for quartz patches.

For Sandveld vegetation types a non-linear model using RATIO and X proved to be the best model (Appendix 3.1). Both X (latitude) and Y (longitude) were significant. However, given the small number of data points only one geographic variable could be used. For both variables z-values showed a distinctly parabolic curve. Although the range in RATIO for Sandveld vegetation types is small in relation to other vegetation categories, in contrast to the quartz patch model, topographic diversity proved to be a significant variable (Appendix 3.1).

Comparing the response of z-values in the three models it is clear that plant biodiversity within different vegetation type categories responds differently in relation to the landscape variables. Although geographic distance and habitat diversity are significant variables explaining the pattern in z-values, this pattern is different in each case. Thus, it was not possible to develop a general model to extrapolate z-values to all Succulent Karoo vegetation types. Therefore, for the remainder of vegetation types that did not fall into one of these three vegetation categories, they were awarded the observed z-value for the geographically nearest vegetation type within their broader vegetation category.

3.3.4 Calculating conservation targets from z-values

Conservation targets were calculated for vegetation types using Equation 3. Examples of the range in conservation target values, expressed as the percentage of vegetation type required to represent a given proportion of plant species occurring in that vegetation type, are calculated for the observed range in z-values (Table 3.3).

If the conservation objective is to represent the majority of biodiversity, say between 70% and 80% of species, within the formal reserve network then the SAR would predict that well in excess of 10% would be required for most land-classes. For the Succulent Karoo vegetation types where a conservative average estimate of the z-value is 0.18 this would translate into a target of between 14 and 30% of the land area required to represent between 70% and 80% of the species respectively.

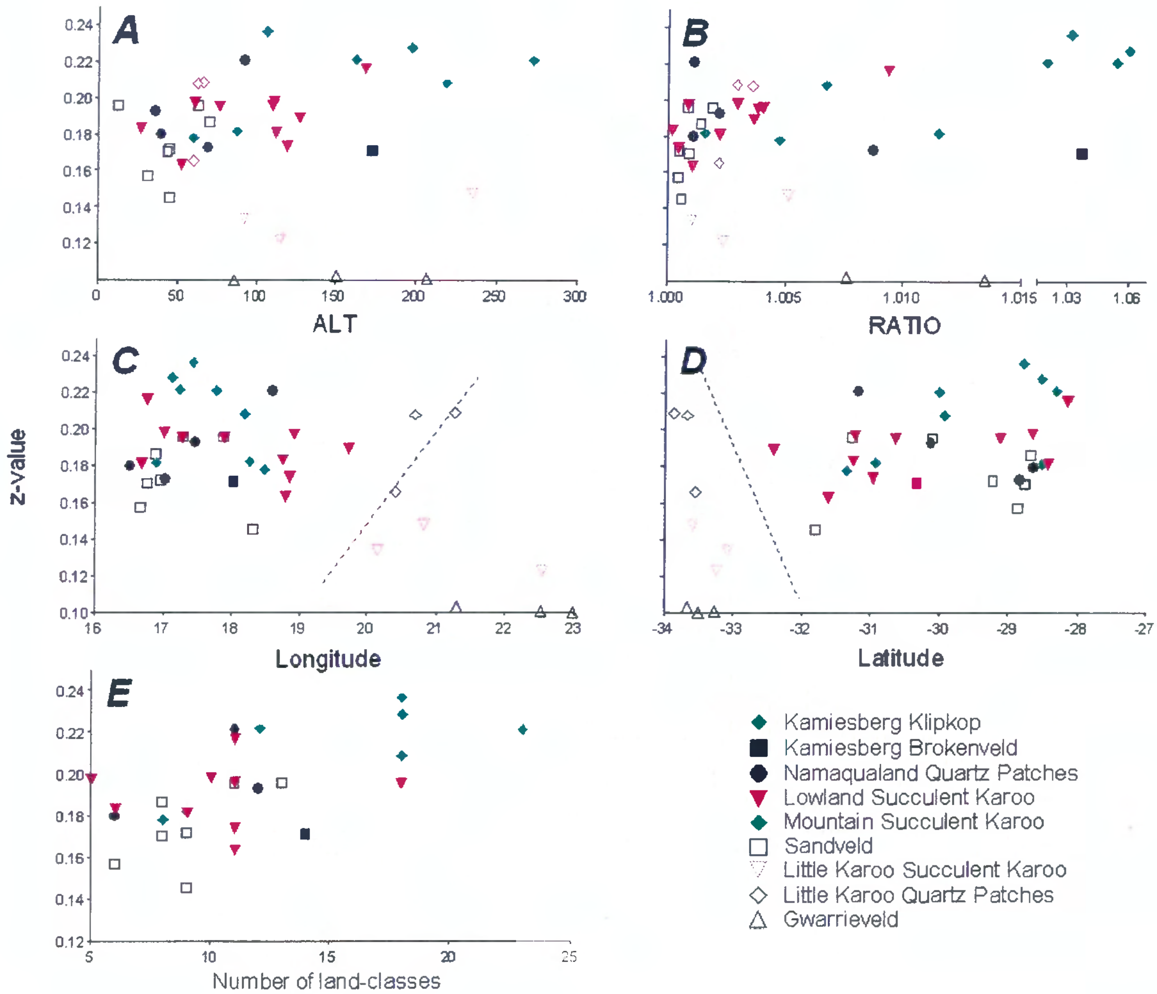


Figure 3.2: The relationship between estimated z-values and: (A) the standard deviation of mean vegetation type altitude (ALT); (B) the ratio between planimetric and surface area of each vegetation type (RATIO); (C) longitude of vegetation type centroid; (D) latitude of vegetation type centroid; and, (E) the number of modelled land-classes per vegetation type. The Namaqualand and Little Karoo regions of the Succulent Karoo are separated by a dashed line in the longitude (C, Namaqualand to the left) and latitude (D, Namaqualand to the right) graphs. In this figure the Namaqualand Succulent Karoo vegetation group has been divided into three categories: Kamiesberg Klipkop, Lowland Succulent Karoo and Mountain Succulent Karoo.

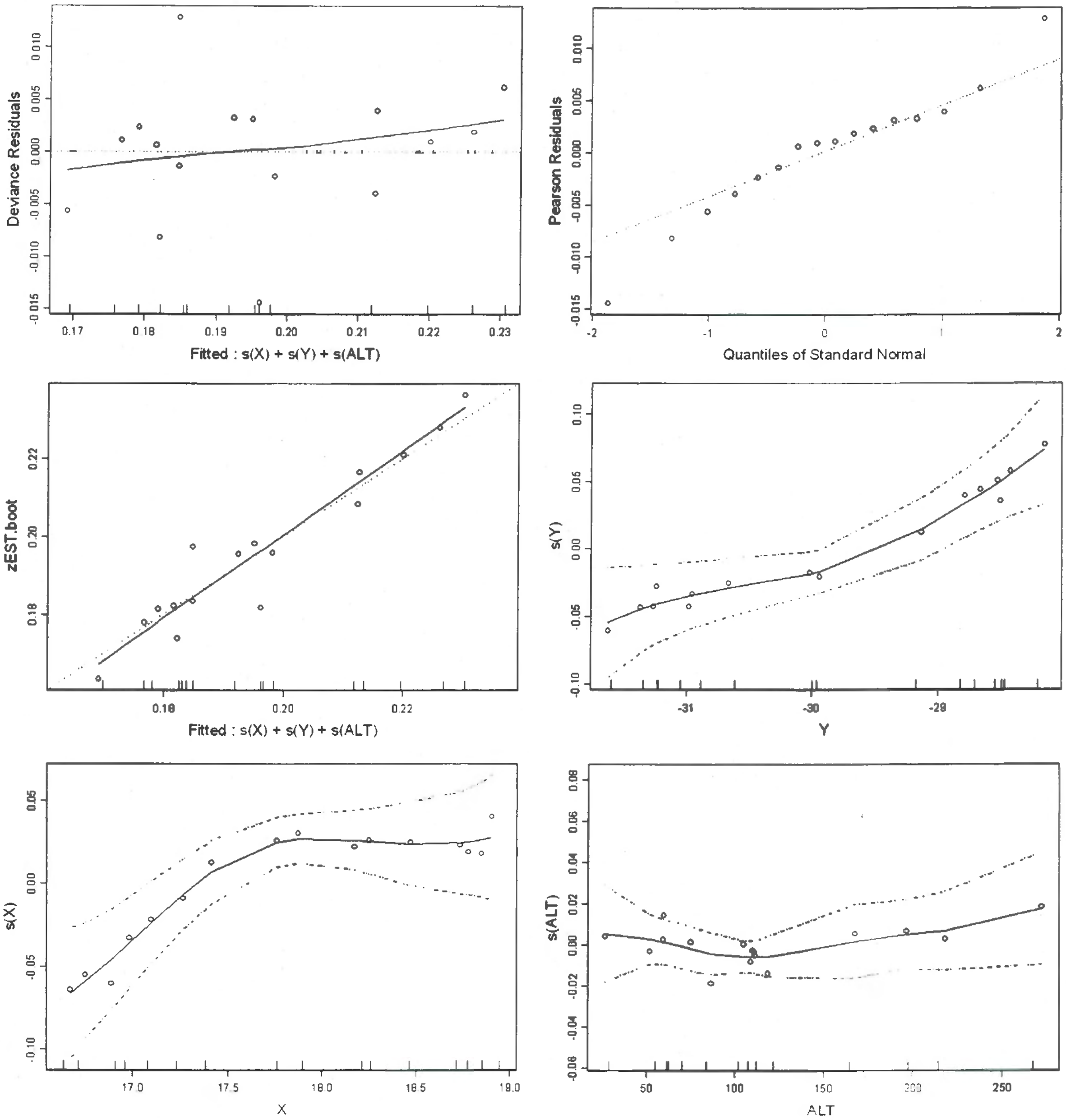


Figure 3.3: Regression model outputs for the Namaqualand Succulent Karoo vegetation types using a generalised additive model (Dispersion Parameter for Gaussian family: 0.0001929; Null Deviance: 0.0065084 on 15 degrees of freedom; Residual Deviance: 0.000578 on 2.999648 degrees of freedom; Linear model: z-value = 1.0632(z-value fitted) – 0.0124; n=15, r^2 = 0.9144; p<0.0001).

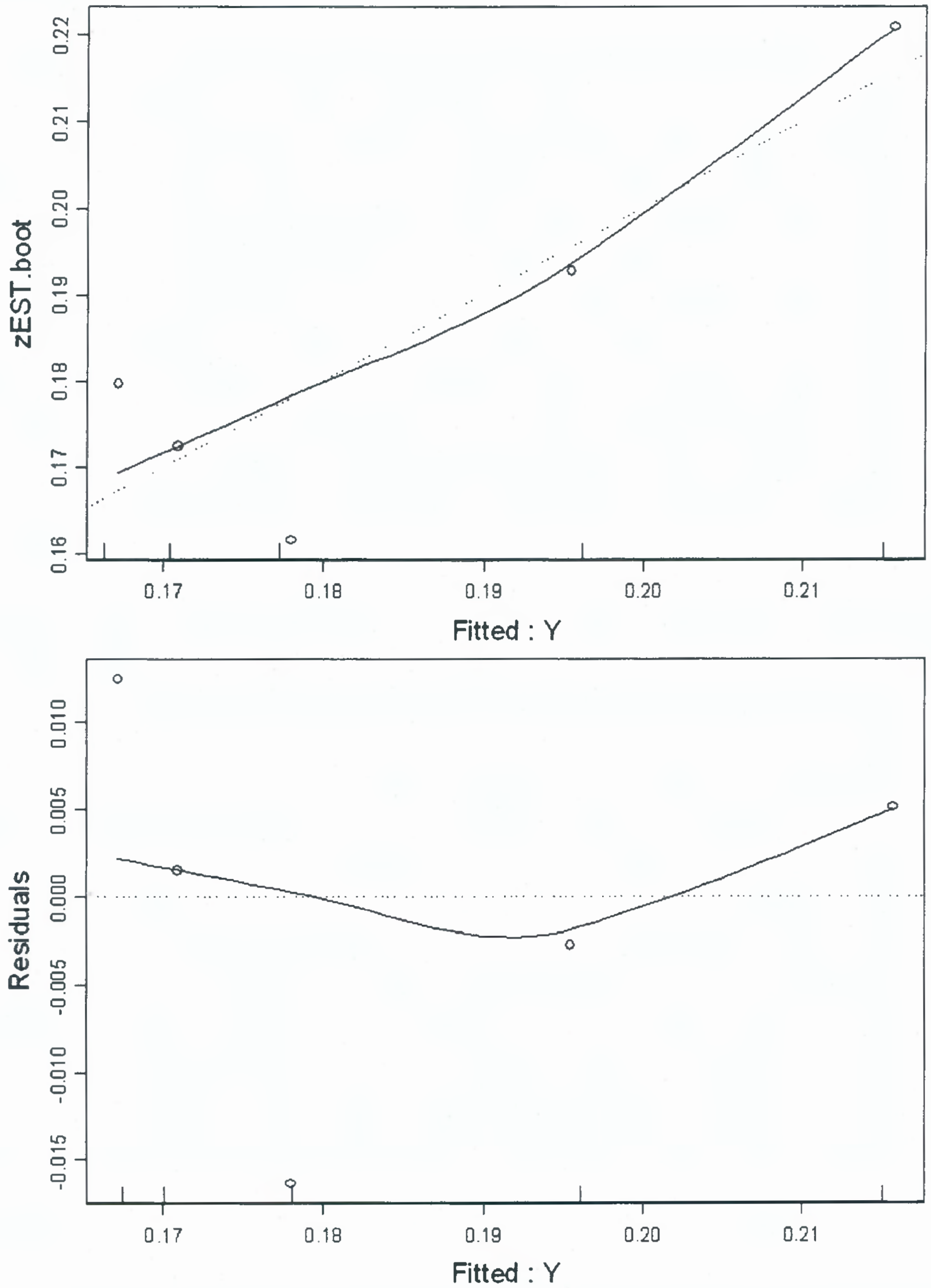


Figure 3.4: The regression residuals (left) and fitted model (right) for Namaqualand quartz patch vegetation types (z-value = -0.0190(Y) - 0.3758; n = 5, r^2 = 0.7792; p = 0.04735).

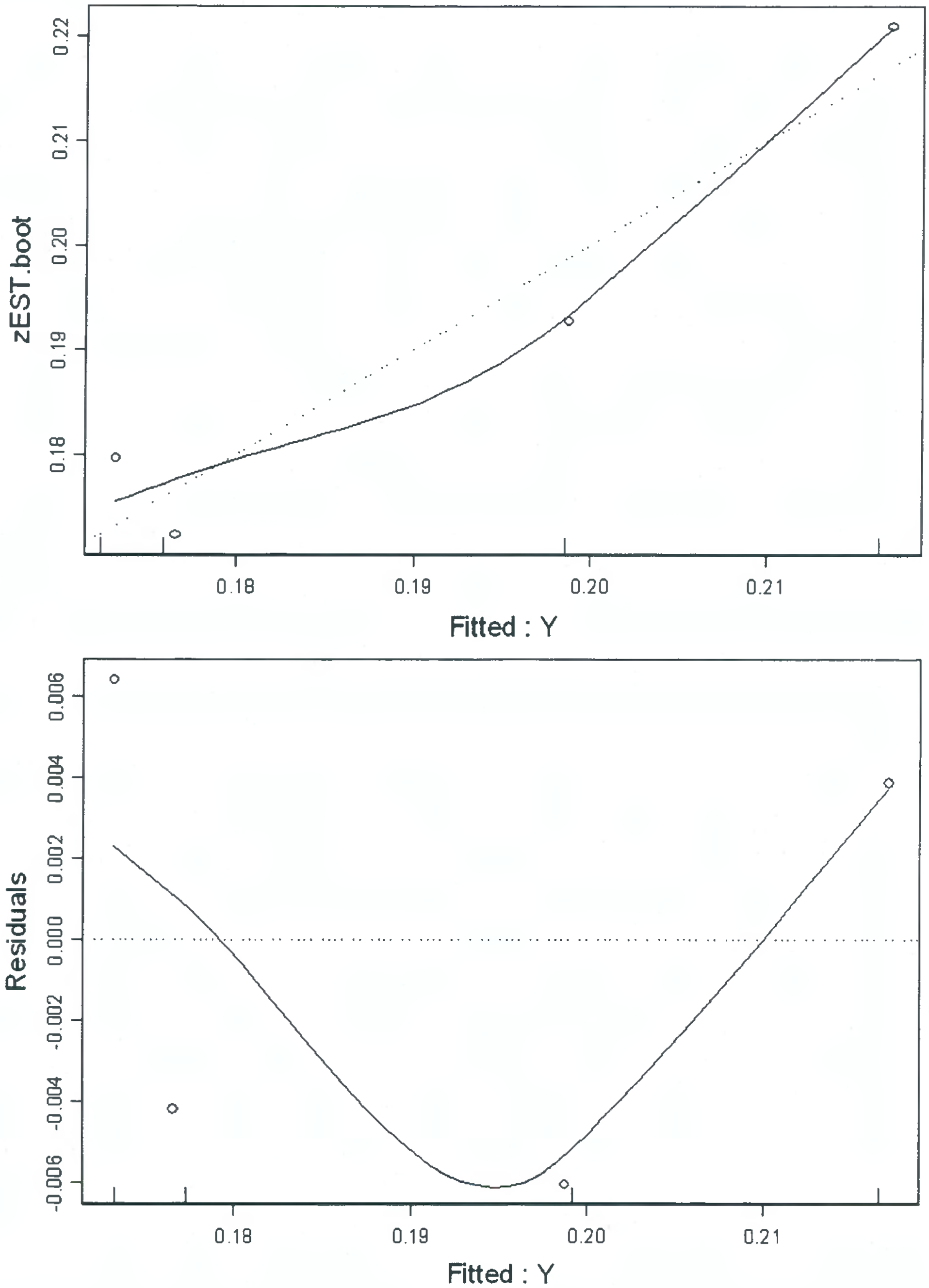


Figure 3.5: The regression residuals (left) and fitted model (right) for Namaqualand quartz patch vegetation types excluding Eastern Bushmanland Quartz And Gravel Patches (z-value = -0.0171(Y) - 0.3162; n = 4; r^2 = 0.9197; p = 0.04097).

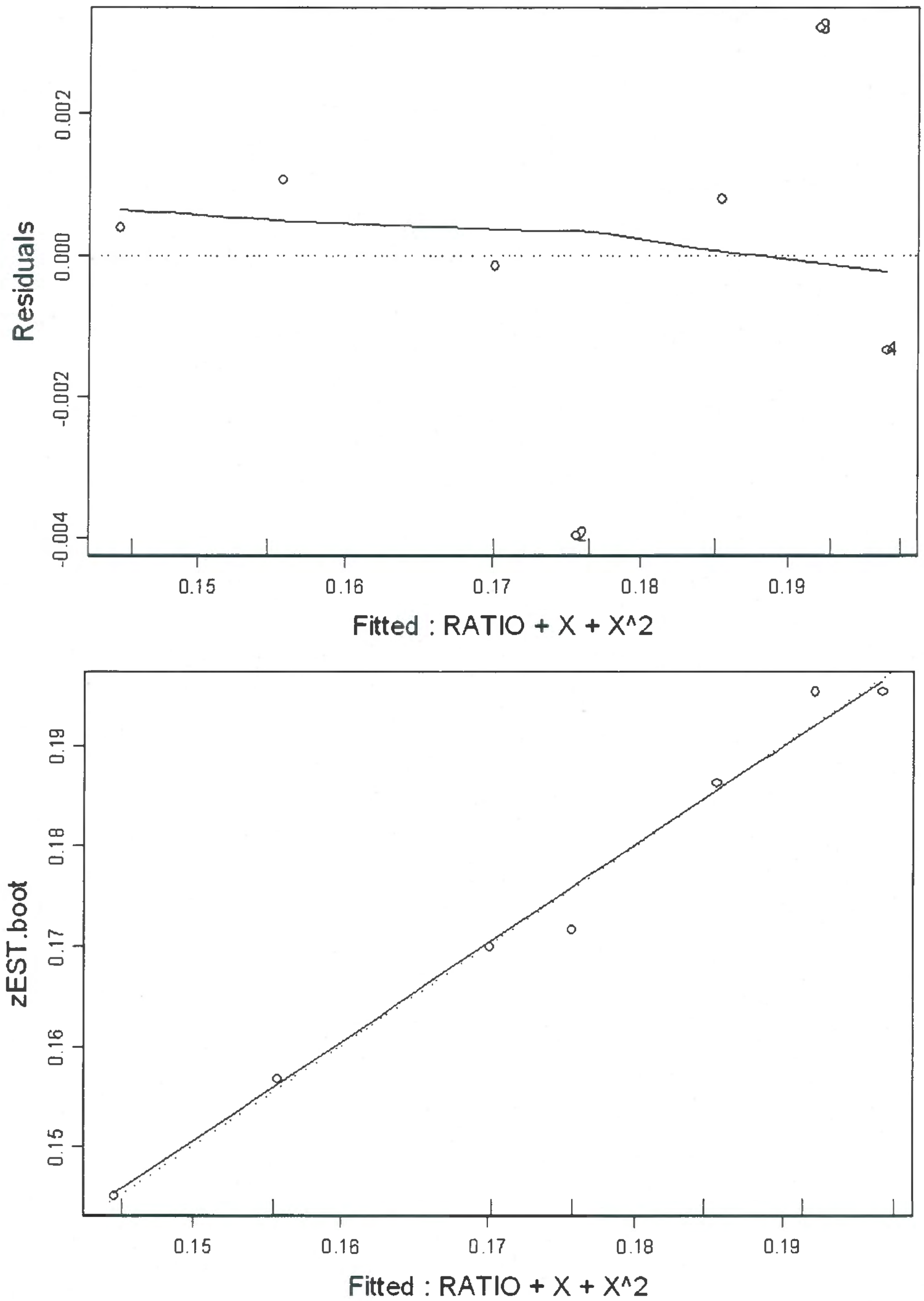


Figure 3.6: The regression residuals (left) and fitted model (right) for Sandveld vegetation types (z-value = 14.7047(RATIO) + 1.9716(X) - 0.0566(X^2) - 31.6955; n=7; R2=0.9866; p=0.002635).

Table 3.3: The percentage of Succulent Karoo vegetation types required to represent a given proportion plant species calculated for the range of observed z-values. Target values are rounded off to the nearest whole number.

		Proportion of species targeted				
		0.5	0.6	0.7	0.8	0.9
z-values	0.1	1	1	3	11	35
	0.125	1	2	6	17	43
	0.15	1	3	9	23	50
	0.175	2	5	13	28	55
	0.2	3	8	17	33	59
	0.225	5	10	20	37	63
	0.25	6	13	24	41	66
	0.3	10	18	30	48	70

3.4 Discussion

3.4.1 Is 10% enough?

The approach to setting targets discussed in this paper does provide an ecological framework for testing the validity of the widely used 10% target. The SAR would predict that for most Succulent Karoo vegetation types a conservation target of 10% of the land area would not be sufficient to conserve that majority of species. It is likely that it is the same for the majority of the land-classes elsewhere on Earth. The 10% target may only be valid for only the most species poor land-classes.

Another important finding of this study is that not all land-classes are equal from a biodiversity perspective. Just as 10% is not enough to represent most species, so applying one target to all land-classes will lead to significant gaps and inefficiencies in any resultant reserve network (Soule 1998; Pressey *et al.* 2003a).

3.4.2 Assumptions of the power model

The approach here assumes that the power model best describes the species-area relationship. This model rests on the assumption that species-abundance distributed in a land-class follows a log-normal distribution (Preston 1948; May 1975; Rosenzweig 1995).

It is questionable whether this assumption holds for most species. It is to be expected that as species-abundance distributions deviate from this distribution so the difference between predicted and actual targets widen. Harte *et. al.* (1999) has proposed an alternative derivation of the power model based on self-similarity in the distribution and abundance of species. This derivation does not assume a log-normal species-abundance distribution. Thus, the power model may still be valid for setting conservation targets if it is not dependent on the species-abundance distribution.

Also, the nature of the power function means that the curve only reaches 100% of species at 100% of area. How valid this is in reality is debateable (Lomolino 2002). It is likely that it would be possible to represent 100% of species within less that 100% of area. Does this represent a breakdown in the validity of the power model as the curve approaches the asymptote or a defect in the model as a whole? This artefact should not stop conservationist from targeting 100% of species, however, the power form of the SAR cannot be used to predict what the actual area will be for achieving this target.

3.4.3 Limitations of using z-values to set targets

The most important limitation of using z-values to set conservation targets is that it says nothing about where species are located in the landscape. It only provides an indication of the rate at which species are likely to be accumulated. Consequently it says nothing about which 20% of the land-class is required to represent the 75% of species being targeted. If species are distributed randomly in a land-class, then reserving any 20% will capture roughly all of the predicted proportion of species targeted. Unfortunately, species are not distributed randomly and it is unlikely that the location of every species in the landscape will be known. Real world applications of these targets will capture larger areas than those predicted by the SAR target due to inefficiencies in adding areas to a conservation network. This stresses the need for at least some species point locality data to help guide conservation decisions.

Secondly, the z-value describes the accumulation of species based on a single occurrence of a species. A target set using z is analogous to saying, "select one occurrence of each species" in a minset. However, it is more than likely that numerous occurrences of common species may actually be incorporated into the notional or real system by the target. This is good since common species often require larger populations for persistence (Lawton 1988). Also, no vegetation types have exclusive species complements; consequently, species will be targeted in many vegetation types. Unfortunately, rare or very patchy (habitat specific) species, i.e. the other 25% of species not targeted, are likely to be missed. Point locality data for species that are good surrogates for this group (e.g. rare habitats) are necessary in the conservation planning process.

If the conservation goal is to select at least three occurrences of each species then the target will have to be increased to accommodate especially rare species. How the SAR method can be used to achieve this needs to be explored.

Lastly, z-values will increase as a result of land-class fragmentation (Rosenzweig 1995). Archipelagos typically have higher z-values than mainland areas. This is generally ascribed to different rates of immigration, extinction and *in situ* speciation that occur in island or naturally fragmented biotas (Diamond and May 1976; Bond *et al.* 1988; Brown and Dinsmore 1988; Rosenzweig 1995; Lomolino 2001b; Haila 2002). Targets derived here assume that a land-class is untransformed. Under anthropogenic transformation, however, a larger area than predicted by the model will be required to achieve the same species target. This is a crucial point that needs to be borne in mind when apply this approach. Species relaxation in fragmented landscape results in a net loss in the original number of species present as species go extinct from habitat patches over time (Brooks *et al.* 1999; Robinson 1999; Debinski and Holt 2000; Gonzalez 2000; Kelt 2001). This effectively increases the z-value as a larger area is required to represent the same given proportion of the original species compliment. How to adapt SAR targets to landscapes under contemporary transformation needs further investigation.

3.4.4 Which estimator?

Probably the largest source of error in this approach lies in the estimation of the true species number for a vegetation type. There is no consensus in the literature as to which is the best estimator to use (Colwell and Coddington 1995; Chiarucci *et al.* 2003; Petersen *et al.* 2003). There is agreement, however, that the Bootstrap estimator is the most conservative (Colwell and Coddington 1995). In the both the model and real-world datasets it was observed that the Bootstrap estimator was consistently the lowest estimate of the seven estimators. Therefore, the estimates of z-values and targets calculated in this paper should to be regarded as conservative and probably underestimates of true targets. The rationale for using the Bootstrap technique here is based solely on the patterns in estimation error for the model datasets. A better approach may be to use the average of several or all of the estimators. Another approach may be to calculate z-values using all estimators and then deriving a target range. This error, however, does not detract from the utility of the SAR for setting targets but is rather a source of error in prediction of the model.

The best means of eliminating this error would be not to use the estimators to calculate the true species number, but instead use an alternative technique for estimating z-values. Two techniques in the literature hold promise in this regard. Firstly, Harte *et al.* (1999) have developed a method of calculating the SAR using species spatial-turnover data. Faith (pers. comm.) has proposed a method for calculating z-values using the environmental diversity index (Faith 2003). Both these techniques can use the same inventory data, but eliminate the need to estimate true species number.

In regions where there are no survey data, but there are inventory data of some form such as museum collections, it would be useful to explore determining z-values directly from the species-abundance distribution in the pooled inventory data for a land-class. Wright (1988) showed that the z-value can be determined directly from this distribution. The assumption here would be that the number of times a species is recorded in an inventory would be indicative of the species relative abundance in the land-class. This would obviate the need for area-based survey sites, and would also create a novel and very important use for museum data.

As a point of clarification, the z-value cannot be calculated directly by generating a species-accumulation curve for a sample of survey sites. This curve is not a species-area curve rather it is a collector's curve. A species-area curve is constructed by adding successively larger sampled areas to the data pool, until one has sampled the entire land-class. The accumulation of species with this progression is then plotted. Survey data are generally sampled at the same spatial scale so simply generating the collector's curve as one randomly adds sites to the data pool will result in significant errors in the z-value. In addition the collector's curve rarely fits the power model.

3.4.5 Further sources of error

Further bias arises as a result of errors in the survey data. Errors in this type of data are a perennial problem. These include sampling biases leading to uneven sampling of land-classes; geo-referencing errors; omission of cryptic species; species identification errors; and, data capture and archival errors. No effort is made here to control for these errors beyond the normal checks and balances, such as checking spelling, involved with collating and curating a large biological database.

Also, the survey data covers a range of projects that span thirty years of research in the Succulent Karoo and involves tens if not hundreds of workers. None of these projects were aimed at landscape-level biodiversity inventory, although phytosociological studies do tend to target all observed plant communities within their respective study areas. The potential for taxonomic errors is high especially as identification of Mesembryanthema (Aizoaceae), the second largest family in the biome, is notoriously difficult (Smith *et al.* 1998). No attempts have been made as yet to estimate the degree of error in this dataset.

Using survey data from a variety of projects that used different relevé sizes is not a significant source of error. Varying the relevé area by an order of magnitude either way (i.e. 10 to 1000m^2) changed the z-value by less than 0.01%. Therefore, knowing the size of the sample relevé is not important to using this technique. Consequently, variable survey area size is not a constraint to using this method. So long as the sample areas are within an order of magnitude of each other they can be combined for the purposes of estimating targets.

These problems highlight the great need for systematic data collection over a variety of scales, to allow for proper comparisons of z-values at local, regional, and global scales. It is imperative that all this is done in a highly standardized manner, and having in mind comparisons at the scale considered. Once such data would be available, it would constitute a great starting point for systematic and biologically meaningful target-setting.

Another source of error lies in the delimitation of land-class boundaries. The vegetation types used here were mapped using expert knowledge. Errors in where the "true" boundaries of the vegetation types lie can lead to over or under estimation of z-values.

For example, the boundaries of the Eastern Bushmanland Quartz and Gravel Patch vegetation type as used in this study are incorrect. In revised versions of the South African vegetation map, this vegetation type has been divided between three vegetation types (one new and two existing) (L. Mucina pers. comm.). The resultant vegetation map not only agrees better with expert assessment, but the vegetation types are also more homogeneous and better reflect landscape-level vegetation patterns. The consequences for the targets are that they will have to be revisited for this area and relevés reassigned to vegetation types according to the new boundaries. The calculated target cannot be extrapolated from the old vegetation type to the new ones. Another problem that arises as a result of incorrect vegetation type boundaries is that relevés get incorrectly assigned to a vegetation type. Such errors can only realistically be controlled through wide expert involvement in the delimitation of land-class boundaries whether using expert mapping or modelling techniques.

3.4.6 Extrapolating z-values

Within a biogeographic province there is considerable variation in z-values. There is, however, a generally agreed strong relationship between species diversity and habitat diversity (Rosenzweig 1995) which was confirmed in this study. Using topographic diversity as a surrogate for habitat diversity, a model was constructed that relates z-values to an independent land-class metric that can be generated from remotely derived GIS data. The advantages of taking this step are substantial. It is now possible to approximate a z-value for all land-classes in the Succulent Karoo based on a measure of the diversity of habitats. Geographic location is important in explaining the pattern in z-values. This may reflect the historical influence of dynamic environments on the evolution of regional floras.

For the Succulent Karoo vegetation types, where there are not enough vegetation types in a category to build a significant model incorporating topographic diversity, the equally strong observed relationships between z-values and geographic location supports the approach of assigning z-values to vegetation types in the same group using the nearest neighbour principle. This approach would hold at least within vegetation groups within a biome.

It is difficult to make predictions for other biomes. It is almost certain that other biomes will have a range in z-values and hence targets. This study makes is very clear that no single target will be suitable for all land-classes within a region. As a very general rule, land-classes with large numbers of range-restricted species will have higher targets. Whether there is a relationship between endemicity or rarity and z-values will need to be determined before more empirical statements can be made. Also, more topographically diverse land-classes will have relatively higher z-values.

3.4.7 General

This work represents the first attempt to quantitatively determine conservation targets for land-classes based on ecological theory. Exciting as this advance is, the limitations of this approach both in terms of input data requirements, data error and model assumptions must not be forgotten.

Conservation practitioners need to also bear in mind that applying SAR targets is only one of many types of conservation targets. This target is based on the hypothesised accumulation of species in a sample of conservation areas. It does not explicitly take into account multiple occurrences of species nor does it tell us anything about where within a land-class the target should be achieved to conserve the target proportion of species. Also, these targets do not tell us anything about requirements for ecological processes. Also, it is important to remember the SAR target does not replace other approaches to setting targets that focus, for example, on minimum viable populations, meta-population dynamics or ecological processes.

From a practical perspective the two major limitations of binary or fixed conservation targets need to be stressed. Firstly, fixed targets distort the effectiveness conservation implementation. Using fixed targets it is possible to achieve targets for land-classes that are just below target, and which probably require minimal effort to achieve, whilst ignoring those land-classes that are far below target and which require significantly more effort to achieve their targets. This approach of picking the low hanging fruits, although not always the case, exaggerates the success of conservation implementation whilst exposing the most vulnerable components, i.e. those least conserved, to potentially greater risk through being sidelined by the implementation process.

Secondly, fixed targets promote the land-use philosophy of "clearing down to target". This is a dangerous philosophy as it is generally accepted that one requires more than just each species represented in a reserve to conserve biodiversity. The SAR target approach applied here does not take into account ecological processes.

Survey plots are little more than slightly-less-than-random-samples of the complete biodiversity present at any point in space. These data, however, have formed the basis of much of the ecological research into how terrestrial systems are structured and work. It is important to make the best use of available information rather than wait to for better data. Biologically informed decisions need to be made regarding conservation action and landscape management. The methodology for setting conservation targets presented here is by no means a "save-all" solution to the problem of setting targets. It should be viewed as a tool that compliments rather than replaces existing empirical or expert based species, population, habitat or ecosystem targets. This method is fraught with methodological and data assumptions that need to be addressed, but in the mean time it would be wise to apply the method mindful of its limitations rather than wait till these problems have been resolved.

It must reiterated that this work does make it clear that the IUCN 10% target is inadequate for capturing the majority of plant diversity within the Succulent Karoo Biome. This trend is probably true for many other terrestrial ecosystems. Further, land-classes are not all equal from a biodiversity perspective and setting a single target for all land-classes does not make good conservation sense.

Appendix 3.1: A summary of model parameters relating z-values to independent landscape variables for the three Succulent Karoo vegetation groups presented in Figure 3.3 to Figure 3.6.

A. Namaqualand Succulent Karoo vegetation types

1. Model using ALT as the topographic variable:

*** Generalized Additive Model ***
Dispersion Parameter for Gaussian family taken to be 0.0001927
Null Deviance: 0.0065084 on 15 degrees of freedom
Residual Deviance: 0.000578 on 2.999648 degrees of freedom
Number of Local Scoring Iterations: 1
DF for Terms and F-values for Nonparametric Effects

	Df	Npar Df	Npar F	Pr(F)
(Intercept)	1			
s(ALT)	1	3	0.695295	0.6137801
s(Y)	1	3	1.467967	0.3800299
s(X)	1	3	5.646122	0.0944951

*** Linear Model ***
Coefficients:

	Value	Std. Error	t value	Pr(>|t|)
(Intercept)	-0.0124	0.0171	-0.7243	0.4808
ALT.fit	1.0632	0.0869	12.2310	0.0000

Residual standard error: 0.006307 on 14 degrees of freedom
Multiple R-Squared: 0.9144
F-statistic: 149.6 on 1 and 14 degrees of freedom, the p-value is 7.32e-009

2. Model using RATIO as the topographic variable:

*** Generalized Additive Model ***
Dispersion Parameter for Gaussian family taken to be 0.0002076
Null Deviance: 0.0065084 on 15 degrees of freedom
Residual Deviance: 0.0006225 on 2.998911 degrees of freedom
Number of Local Scoring Iterations: 1
DF for Terms and F-values for Nonparametric Effects

a	Df	Npar Df	Npar F	Pr(F)
(Intercept)	1			
s(Y)	1	3	0.4030767	0.7624483
s(RATIO)	1	3	0.8653086	0.5459514
s(X)	1	3	0.8910422	0.5367138

*** Linear Model ***
Coefficients:

	Value	Std. Error	t value	Pr(>|t|)
(Intercept)	-0.0065	0.0176	-0.3699	0.7170
RATIO.fit	1.0332	0.0893	11.5677	0.0000

Residual standard error: 0.006636 on 14 degrees of freedom
Multiple R-Squared: 0.9053
F-statistic: 133.8 on 1 and 14 degrees of freedom, the p-value is 1.496e-008

B. Sandveld vegetation types

*** Linear Model ***
Coefficients:

	Value	Std. Error	t value	Pr(>|t|)
(Intercept)	-31.6955	2.5050	-12.6529	0.0011
RATIO	14.7047	2.9038	5.0639	0.0149
X	1.9716	0.2152	9.1615	0.0027
I(X^2)	-0.0566	0.0061	-9.2076	0.0027

Residual standard error: 0.00315 on 3 degrees of freedom
Multiple R-Squared: 0.9866
F-statistic: 73.4 on 3 and 3 degrees of freedom, the p-value is 0.002635

C. Namaqualand Quartz Patch vegetation types

1. Model including the Eastern Bushmanland Inselberg and Quartz Patch vegetation type

*** Linear Model ***
Coefficients:

	Value	Std. Error	t value	Pr(>|t|)
(Intercept)	-0.3758	0.1726	-2.1775	0.1176
Y	-0.0190	0.0058	-3.2539	0.0474

Residual standard error: 0.01238 on 3 degrees of freedom
Multiple R-Squared: 0.7792
F-statistic: 10.59 on 1 and 3 degrees of freedom, the p-value is 0.04735

2. Model with only the four strict Namaqualand Quartz Patch vegetation types

*** Linear Model ***
Coefficients:

a	Value	Std. Error	t value	Pr(>|t|)
(Intercept)	-0.3162	0.1061	-2.9803	0.0966
Y	-0.0171	0.0036	-4.7874	0.0410

Residual standard error: 0.007422 on 2 degrees of freedom
Multiple R-Squared: 0.9197
F-statistic: 22.92 on 1 and 2 degrees of freedom, the p-value is 0.04097

4 Targeting ecological processes - A top down approach

The previous chapter introduced the concept of targets in conservation planning and presented a novel method for estimating biodiversity pattern targets. This chapter focuses on the second component of conservation targets - biodiversity or ecological process targets. Developing a comprehensive set of process targets based on empirical data of a selected suite of processes in beyond the scope of this study. This reality probably reflects the real world situation that most regional conservation planning exercises find themselves in. Unless there is existing research that specifically identifies the spatial components of ecological processes, it is unlikely that this research will be done specifically for any particular planning exercise. This lack of information does not mean, however, that targets for processes should not be included in the planning process.

Thus, a challenge facing this project and more broadly conservation biology is to integrate the available information on the spatial requirements of ecological processes to develop generalization or at least guidelines that can be applied to ecosystems where there is little data on such processes. Given the global threat facing biodiversity it is imperative that at least an attempt is made. It is better to make conservation decisions based on the perceived understanding of an ecosystem than make no decisions citing the lack of hard scientific data. This indecision will not stop or influence the agents of landscape transformation and biodiversity loss.

Conserving ecological processes invariably requires the conservation of a significantly larger proportion of the landscape than is required to represent biodiversity pattern (Cowling *et al.* 1999a; Soulé and Terborgh 1999; Pressey *et al.* 2003a). Conserving ecological processes means not only conserving the area where a species currently occurs, but also sufficient of its habitat so that it is able to continue with the day-to-day task of survival, now and into the future. From the planning perspective this involves determining the minimum amount of area required to conserve these ecological processes.

Ecological processes include both biological (e.g. survival, reproduction, dispersal and interaction) and abiotic (e.g. geomorphological, pedological and hydrological) processes.

Conservation planning is concerned about the spatial requirements of these processes. Thus, conservation efforts aimed at conserving ecological and evolutionary processes require biologists to integrate understanding of patterns and processes of landscape change, such as habitat loss and fragmentation, with detailed responses of individual populations and species to these broad-scale modifications (Collinge 2001)

There is a growing literature in the field of spatial ecology that addresses the spatial requirements of the biological component of ecological processes. Spatial ecology centers on how a landscape's spatial configuration influences the population and community dynamics of organisms; and, has emerged in the last decade out of landscape, population and community ecology (Collinge 2001)

Research into the biological impacts of habitat loss or fragmentation addresses the question of the spatial requirements of ecological processes, and indirectly what meaningful process targets should be. Habitat fragmentation implies loss of habitat, reduced patch size and an increasing distance between patches, and also an increase in new habitat (e.g. agricultural fields) (Hanski 1991; Andren 1994). Fragmentation in natural systems also occurs through natural agents such as fire or tree falls. However, the largest-scale cause of habitat fragmentation is the expansion and intensification of human land-use (Andren 1994)

The impacts of fragmentation for native populations may occur along a continuum from devastating to relatively benign (Collinge 2001). For example, the response of species can vary according to the species considered; its life history; mobility; spatial requirements; vulnerability to habitat edges; the character of the landscape interspersed with preferred habitat; or, the spatial configuration of the preferred habitat (Collinge 2001). Generally though, a decrease in the amount of available habitat results in a decrease in population sizes and a loss of species from the system (Parker and Mac Nally 2002). This is a result of a decrease in the amount of habitat available to organisms and a gradual break down in ecological processes. Both the species-area relation and the random-sample hypothesis have been proposed as models to describe the loss of species from shrinking landscapes (Andren 1999). At some point, however, the decline in a population reaches a persistence threshold where the population crashes in response to a very small change in the amount of habitat (Figure 4.1a). Two broad persistence thresholds are recognized in the literature, defined according to the causal factors – a fragmentation threshold and an extinction threshold.

A fragmentation threshold occurs as a result of the effect of the spatial arrangement of habitat patches on ecological processes. Below this persistence threshold, indicated by a rapid decline in the probability of the landscape to support viable populations, habitat arrangement becomes important in determining the persistence of populations (Flather and Bevers 2002). The identification of this threshold emerged from percolation theory and the resultant neutral landscape models developed with reference to the flow of liquids through lattices of material aggregates (Wiens *et al.* 1997). Percolation theory provides a neutral model against which to test alternative hypotheses about how landscape structure affects the abundance, distribution and behavior or organisms (McIntyre and Wiens 2000). In ecology these models have been used to model the movement of individuals or the spread of disturbances through landscapes comprising blocks of suitable and unsuitable habitat (Green 1994; Wiens *et al.* 1997).

The extinction threshold is the minimum amount of habitat required for a population of a particular species to persist in the landscape (Fahrig 2002). Extinction thresholds are characterized by abrupt declines in the patch occupancy of a metapopulation across a narrow range of habitat loss (With and King 1999b), and at the amount of habitat at which mortality balances reproduction over the landscape (Fahrig 2002) (Figure 4.1c). This threshold emerged from metapopulation theory and is the threshold below which a population is likely to go extinct. Both threshold types may be characterized by similar patterns in population decline; however, the extinction threshold is a result of change in intrinsic population demographic properties in response to habitat amount and structure, and not a direct result of landscape structure or permeability.

Most fragmentation studies agree that habitat amount accounts for almost all of the variation in observed population size in fragmented landscapes (Fahrig 2002; Flather and Bevers 2002). Only at low habitat amounts (i.e. below the fragmentation threshold), does arrangement become important as population persistence becomes more uncertain due to the increase in migration/dispersal mortality (Flather and Bevers 2002) (Figure 4.1d and Figure 4.2). This threshold is dependent on how habitat is interpreted in the landscape, especially the suitability of the matrix to support individuals. Consequently, the threshold may be very difficult to detect in natural systems (Flather and Bevers 2002). Predicting the fragmentation threshold varies also depending on the model used (Fahrig 2002) and the variables considered (Flather and Bevers 2002) (Table 4.1). Variables include habitat amount; arrangement; patch size; edge length; arrangement (isolation); proximity of patches to larger patches (Fahrig 2002; Flather and Bevers 2002); habitat suitability or quality (Cane 2001); and, habitat spatio-temporal dynamics (Keymer *et al.* 2000).

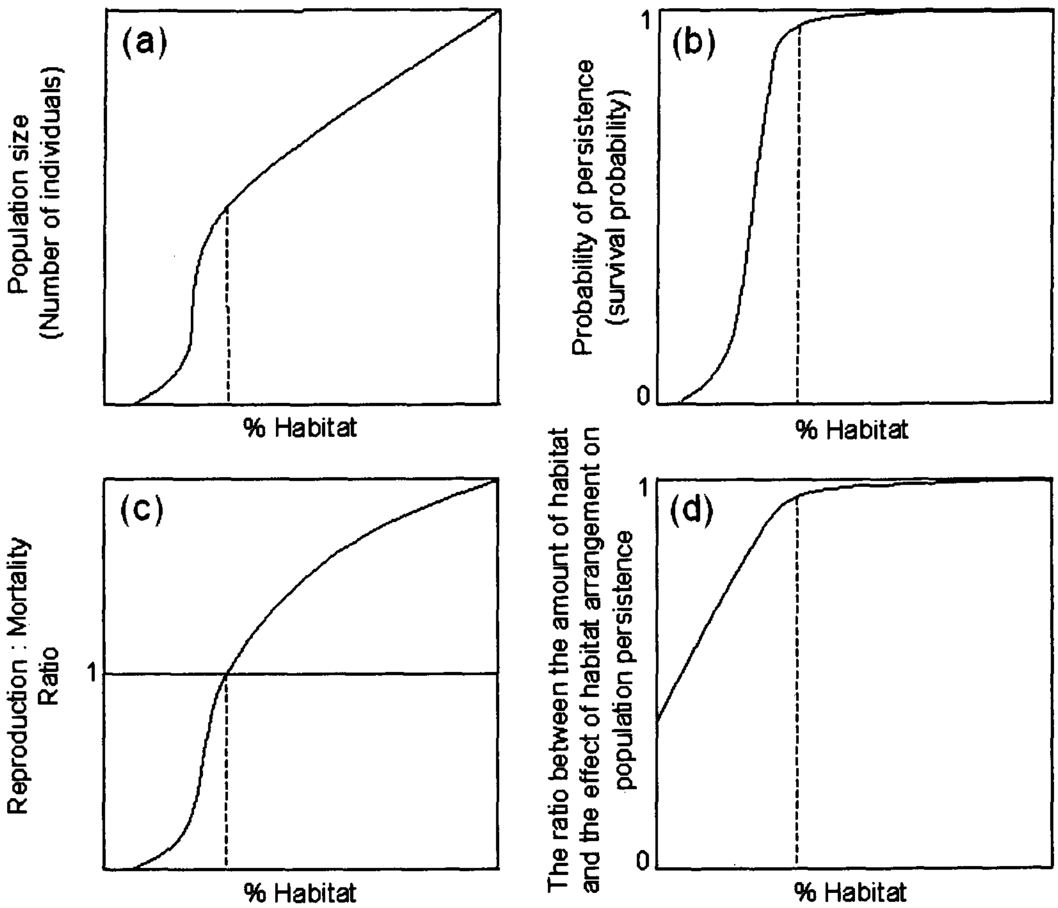


Figure 4.1 (a-c) Different ways of illustrating persistence thresholds that arise as a result of habitat loss. The vertical dashed line indicates the approximate percentage of remaining habitat (% habitat) at which the threshold occurs. (d) An interpretation of the fragmentation type persistence threshold illustrated by the proportional effect of habitat amount versus habitat arrangement in determining the persistence of a population in relation to habitat loss. Only below the fragmentation threshold does habitat arrangement begin to exert a significant effect on population persistence.

For organisms that are resistant to habitat amount effects there does appear to be a landscape structural threshold in lacunarity (a measure of inter-patch distance) at about 20% that affects behavior and potentially persistence (With *et al.* 2002). This threshold has been demonstrated in a number of empirical and simulation studies (Andren 1994; Green 1994; With and Crist 1995; With *et al.* 1999; With and King 1999a; With and King 1999b; With and King 2001). However, different habitat threshold values emerge depending upon whether the effects of landscape structure are being assessed on search behaviors, distribution patterns, population persistence, predator-prey interactions or communities (Lande 1987; Tilman *et al.* 1994; Kareiva and Wennergren 1995; With and Crist 1995; Bascompte and Sole 1998; With and King 1999a; With and King 1999b; Ferreras 2001; Bissonette and Storch 2002).

Generally, the extinction threshold is much higher than the fragmentation threshold as demographic processes tend to break down long before landscape permeability factors influence population persistence (Figure 4.2). Simulation models show that demographic processes: reproduction, dispersal and survival especially in the matrix, i.e. outside of a species preferred habitat in transformed habitats, are more important than fragmentation in determining survival probability (Fahrig 2001; Gibbs 2001).

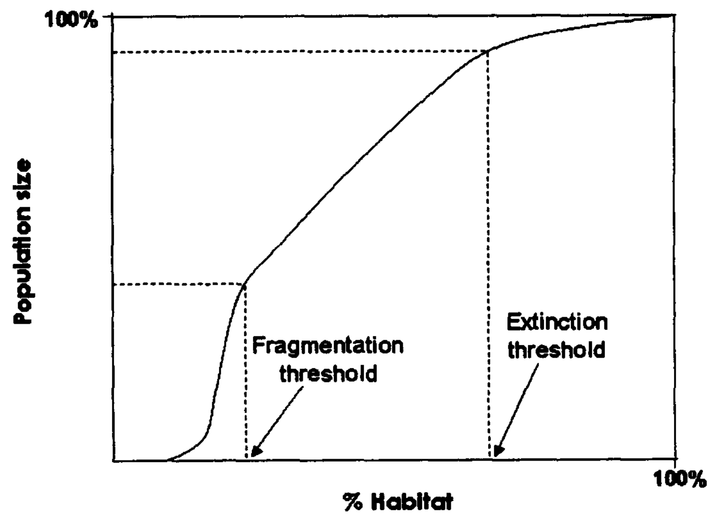


Figure 4.2 A conceptual framework illustrating where fragmentation and extinction thresholds occur in relation to habitat loss.

There are some useful generalizations regarding persistence thresholds that can be drawn from the fragmentation literature that could be incorporated into conservation planning:

- It is important not to confuse the extinction and fragmentation thresholds. The extinction threshold is the minimum amount of habitat below which the population goes extinct, whereas the fragmentation threshold is the amount of habitat below which habitat fragmentation (i.e. habitat pattern) may affect population persistence. While the extinction threshold can occur across a wide range of habitat amounts (20-75%) (Fahrig 2001), fragmentation thresholds appear to occur at about 20-30% of habitat remaining (Andren 1994).
- There is no single magic threshold value. Thresholds will vary depending on the organism concerned; the nature of the landscape; and, the biological permeability of the new habitat created as a result of habitat loss.
- Landscape connectivity must be defined relative to the patch-specific movement patterns of organisms. It is not possible to predict from a land-cover map alone whether or not a landscape will be fragmented or connected for particular species guilds (Wiens *et al.* 1997).
- In addition, it is not possible to deduce a threshold based on landscape metrics alone, as the effect of habitat amount on biological processes is generally more important in determining thresholds. Attempts have been made to develop more ecologically scaled landscape indices that incorporate metapopulation variables with landscape structural metrics (e.g. Vos *et al.* 2001)
- From a conservation perspective, a threshold is the point of impending population crash. It is important to halt habitat loss long before this point if extinction is to be avoided.
- Thresholds are difficult to predict from observational data, as population demographics may appear "normal" up to the point of the threshold. At the threshold it may be too late for populations to avert inevitable extinction.
- Information on movement rates of organisms appears to be the most important variable in predicting extinction thresholds (Fahrig 2001).
- Where inter-patch distance cannot be altered the best practice is to improve the quality of the matrix, in other words improving the permeability of the matrix can reduce the persistence threshold (Ferreras 2001). Restoration of degraded areas or application of organic farming practices that are more biodiversity friendly are required in landscapes that are at or below critical habitat thresholds in order to improve survival of dispersing individuals in the matrix.

- Below the fragmentation threshold adding habitat to the biggest patch in the landscape is most important for population persistence. Above this threshold simply adding area anywhere is important (Flather and Bevers 2002).
- Critical thresholds do exist (Andren 1999; Monkkonen and Reunanen 1999). The most important management implication of their existence is that biological diversity is not a linear function of landscape composition. Relatively small amounts of habitat loss may result in a major impoverishment of diversity (Monkkonen and Reunanen 1999).

The exploration of these thresholds provides useful insights into the functioning of natural populations. More importantly, though, they provide a potential means to determine a biologically meaningful ecological process target for landscapes. All papers examined point out that it is not possible to generalize persistence thresholds for organisms. This is generally confirmed by studies of species richness in relation to habitat area, in which estimates of percent natural habitat required for persistence of all species in an area ranges widely from 20 to 75% (Margules and Nichols 1988; Saetersdal *et al.* 1993; Soule 1998). Unfortunately, conservation planning requires that for broad scale landscape planning generalizations are going to have to be made.

It can be argued, however, that it is indeed possible to make generalizations for landscapes based on these findings by planning for those organisms most sensitive to habitat loss. Liebig's law of the minimum that states that "when several factors are involved in the development of an organism and one is available in only small quantities, that single factor will determine the organisms' success or failure". For example, if data were unlimited it could be possible to estimate the persistence threshold for all organisms in a landscape based on a series of metapopulation-based neutral landscape models. These thresholds could be either of the extinction or fragmentation type depending on the organism concerned and could range from say 5 to 55% of landscape required for persistence. If the conservation goal were to preserve sufficient space for ecological processes such that all species in the landscape were able to persist, then the landscape conservation target would have to be 55% of the landscape. This, however, is not enough. It is not sufficient to target the threshold, as this is the point of impending population crash. It is important to stop habitat loss prior to reaching this point. Thus, an ecologically more meaningful target for a landscape may be the maximum observed threshold plus a buffer amount of 5 or 10%. In formulating a landscape level ecological process target it makes sense to focus on those species most sensitive to habitat loss. From this perspective, such species could be termed "focal" species (Lambeck 1997a; Bunn *et al.* 2000; Carroll 2001; Kintsch and Urban 2002; Lindenmayer *et al.* 2002).

For Succulent Karoo, and probably most of the world outside Europe and North America, presently there is no time, information or resources to develop an empirical approach to setting landscape targets such as described above. In the short term, planning will have to rely on available insights from other systems and make conservative estimates of what are likely to be realistic landscape targets. A review of available literature (Table 4.1) suggests that a conservative landscape target that focuses on the most sensitive species in a landscape will be determined by extinction and not fragmentation thresholds and should lie between 50-70% for most landscapes.

Any landscape or ecological process conservation target framed in this context can be defined as the minimum amount of natural habitat that must to remain in the landscape in order to ensure the long-term survival of the majority of species. Alternatively, it can be stated as being the average extinction threshold for the group of most transformation-sensitive species inhabiting the planning domain.

This ecological process target does not imply that 50 or 70% of the landscape must be in formal reserves, nor does it advocate that the remaining 40% of the landscape can be transformed. Based on ecological observations and theory, this is a minimum extent of natural habitat that planners must strive towards retaining in the landscape if they wish to conserve the majority of ecological processes. It can also be used as a benchmark with which to assess the impact of broad-scale land-use planning scenarios. Thus retained habitat can be divided amongst a number of land-use types that do not involve the loss of natural habitat or permanent interruption of ecological processes

In the absence of physical data on the spatial requirements of ecological processes, the ecological literature does provide some insights that present a theoretical and empirical foundation for developing generalized ecological process targets for landscapes. Both percolation (landscape fragmentation) and metapopulation theory provide a basis for the prediction of critical thresholds in habitat area and structure below which populations would be expected to go extinct in the landscape. These thresholds should provide the basis for landscape targets in conservation planning.

Table 4.1 Examples of extinction and fragmentation thresholds discussed in the literature.

Source	Organism	Type of Threshold	Threshold (% of natural habitat remaining)	Method of deriving threshold
(Fahrig 2001)	-	Extinction	Mean approx. 40-65% (range 1-99%) below which survival probability increases significantly	General, stochastic, individual-based, spatially explicit model of population dynamics and movement of a hypothetical organism in a hypothetical landscape examining the effect of four predictor variable – reproductive rate, dispersal rate, matrix quality and habitat pattern
(Flather and Bevers 2002)	-	Extinction	30-50%	Reaction diffusion models
(Green 1994)	Fire	Extinction	Ca. 84% of landscape unburnt (viz. if fire burns >=14% of landscape at once then risk of species extinction >0)	Percolation theory-based cellular automata models
(Green 1994)	Gene-flow	Extinction	40-60% (viz. below this critical threshold genetic divergence favored over homogenization)	Percolation theory-based cellular automata models
(Vos *et al.* 2001)	Birds	Extinction	Metapopulation viability threshold: <45% occupied patches = non-viable population & >60% occupied patches = viable population	Population viability simulations using METAPHOR metapopulation model calibrated for 2 bird species
(With and King 1999b)	-	Extinction	Extinction threshold rage from 1-99% depending on parameters Poorly dispersed organisms: 60-70% Moderately dispersed: 10-20% Well dispersed: <10%	Metapopulation model combined with a neutral landscape model and a fractal habitat distribution comparing model organisms with a range of reproductive, dispersal and patch occupancy potentials
(Heinen *et al.* 1998)	Small mammals	Fragmentation	30% a critical threshold below which populations have higher risk of extinction	"Model chipmunks" with a high dependence on native habitat
(Andren 1994)	Birds and mammals	Fragmentation	Mean 20% (range 3-60%)	Summary of empirical observation data
(Fahrig 1997)	-	Fragmentation	=<20%	General, stochastic, individual-based, spatially

				explicit model of population dynamics and movement of a hypothetical organism in a hypothetical landscape
(In Wiens *et al.* 1997) summary	-	Fragmentation	25-59% critical threshold in landscape permeability	Simple neutral landscape models with differing combinations of habitat clustering, habitat permeability and organism movement
(Wiens *et al.* 1997)	Insects	Fragmentation	20% critical threshold before habitat configuration begins to influence insect behavior	Neutral landscape model-type experimental model system
(With and King 2001)	Birds	Fragmentation	20% a critical landscape structure threshold for edge sensitive species	Metapopulation model combined with a neutral landscape model and a fractal habitat distribution to explore the relationship between habitat structure and demography
(With *et al.* 2002)	Insects	Fragmentation	<20% habitat results in breakdown of interaction	Neutral landscape model experimental model system using aphids and predatory coccinellid bugs demonstrating break down on predator-prey interactions
(Cox and Engstrom 2001)	Birds	No threshold sought	Mean 52% (range 15-75%). Area required for extinction probability to be below 0.1.	Empirical demographic data. Simulations performed with a stochastic, stage-structured metapopulation model to determine extinction probability for different conservation easement scenarios

5 Designing a core reserve in the Knersvlakte

5.1 Introduction

In 1999, Desmet *et al.* produced a report for the Leslie Hill Succulent Karoo Trust (LHSKT) detailing a spatial framework for the development of a conservation area in the Knersvlakte aimed at conserving the region's unique floral biodiversity. Since the completion of this project, a number of important developments have taken place that warrant the re-visiting of the initial planning outcomes.

Firstly, the lead conservation agent responsible for implementing the reserve has changed from the South African National Parks (SANParks) to the Western Cape Nature Conservation Board (WCNCB). The constraints that the SANP placed on the location and size of the reserve thus fall away allowing for the planning of a reserve unconstrained by human infrastructure or a limit on size. Secondly, in April 2002, the West Coast District Municipality, with support from the Western Cape Provincial Government, adopted the biosphere reserve model as a basis for regional land-use planning by appointing Dennis Moss and Associates to draw up the Knersvlakte Bioregion Spatial Plan (Anon. 2002). Consequently, WCNCB recently expressed the desire to have the reserve re-examined within the context of a broader biosphere reserve plan for the region (K. Hamman, pers. comm.). Thirdly, in 1999 a hiatus in reserve development followed the acquisition of Moedverlooren, a farm located in the core area of the proposed new reserve.

Recently, both the LHSKT and WCNCB have shown a renewed interest in establishing this reserve and in November 2002 this project was initiated on a request from the LHSKT. This in part was precipitated by the fact that the Knersvlakte has been identified as a regional conservation priority in the Succulent Karoo Ecosystem Plan (SKEP) project (Driver *et al.* 2003b). An additional request made by the WCNCB asked that the reserve be nested within a broader biosphere reserve framework that could be tied in with Board's regional conservation plans as well as the bioregional spatial planning processes being conducted by the Dennis Moss Partnership. The biosphere reserve is not strictly a conservation vehicle. It is mechanism that integrates land-use and conservation needs into a single planning framework. The aim of the bioregional planning philosophy is to promote sustainable development through combining these traditionally separate planning frameworks (Anon 2003a). Thus, the products of this study are intended for both land-use and conservation planners.

5.1.1 Objectives of this study

The objectives of this study are two fold. Firstly, to revisit the previous reserve layout and using a similar systematic conservation-planning protocol, examine the options and priorities for the creation of a core reserve in the absence of the constraints placed on the initial layout. A strong emphasis has been placed on the conservation of the patterns of biodiversity and ecological and evolutionary processes associated with the unique habitats and biota of the region, namely the quartz fields, limestone and quartzite rocky habitats. These habitats support the overwhelming number of the Knersvlakte's endemic plant species.

The secondly objective of this study was to nest the core reserve within the context of a broader regional biosphere reserve comprising a network of connected core, buffer and corridor conservation compatible land-use areas. The design of the biosphere reserve is introduced and discussed in the following chapter.

By way of introduction, the following sections discuss briefly the plant diversity of the Succulent Karoo and Knersvlakte, and the systematic conservation-planning protocol used in this study

5.1.2 The Succulent Karoo

The Knersvlakte comprises one of 12 bioregions (Hilton-Taylor 1994a) identified for southern Africa's succulent karoo biome. This biome is a predominantly winter-rainfall desert region that occupies 112 000 km^2 on the arid fringes of the Cape Floristic Region. On account of its spectacular biodiversity, this region is the only arid land to qualify as a global biodiversity hot-spot (Cowling and Pierce 1999). It includes 6356 species of plants (26% endemic and 14% near-endemic) and is home to the richest succulent flora in the world (Driver *et al.* 2003b). It is also a centre of diversity for reptiles and many different groups of invertebrates. The recent and explosive diversification in the Mesembryanthemaceae, the largest succulent plant family in the region, has been described as an event unrivalled among flowering plants (Ihlenfeldt 1994; Desmet *et al.* 1998; Klak *et al.* 2004).

As a consequence of the unusual composition and high endemism, the flora of the Succulent Karoo is unique (Cowling and Hilton-Taylor 1999). Local and regional plant richness is very high with an average of 70 species, and as many as 113, being recorded in a tenth-hectare plot (Cowling *et al.* 1998). Larger areas support about four times the number of species than comparable winter-rainfall deserts elsewhere in the world (Cowling *et al.* 1998). This high regional richness is the result of high compositional change of species-rich communities along environmental and geographical gradients, i.e. high beta and gamma diversity, respectively (Cowling and Hilton-Taylor 1999). Many species are extreme habitat (mainly edaphic) specialists of limited range size. Point endemism is most pronounced among succulents (especially Aizoaceae (Mesembryanthema)) and bulbous lineages, and is concentrated on hard substrata, especially quartzite and shale koppie and quartz lag-gravel plains (Schmiedel and Jürgens 1999; Schmiedel 2002). The Succulent Karoo is home to 952 Red Data Book species with 25% known from only one or two quarter degree squares (i.e. 136 000 ha), and with 32% endemic and 26% near-endemic to the biome (Driver *et al.* 2003b).

Given its global significance as a biodiversity hot-spot (Cowling and Pierce 1999), and its long-standing recognition as a regional conservation priority (Rebelo and Siegfried 1992a; Hilton-Taylor 1994a), the current protected area system in the Succulent Karoo is woefully inadequate. Only 3.5% of the Succulent Karoo biome falls into statutory reserves managed primarily for biodiversity conservation such as National Parks and Provincial Nature Reserves, and a further 2.3% in statutory and non-statutory reserves managed for biodiversity conservation and/or other land uses such as the Richtersveld National Park, municipal reserves and conservancies (Driver *et al.* 2003b). Hilton-Taylor (1994a) reported that 2.1% of the biome was conserved in statutory reserves, indicating a growth of nearly 60% in formal reserves in the biome over the last 10 years. Despite this growth, the overall conservation estate (statutory and non-statutory reserves = 5.8%) in the biome is still far below currently accepted minimum of 10% recognised by conservationists (Anon 2003b).

More than 90% of the Succulent Karoo is used as natural grazing (Hilton-Taylor 1994a), a form of land use that is, at least in theory, not incompatible with the maintenance of biodiversity and ecosystem processes. About 100 000 km^2 remains in a natural or semi-natural state. However, much of this remaining natural habitat is vulnerable to a wide range of land-uses (Cowling *et al.* 1999a). These, in order of their overall importance, are:

- the expansion of communally-owned land and the associated overgrazing and desertification;
- overgrazing of commercial (privately-owned) rangelands;
- cropping agriculture, especially in the valleys of perennial rivers;
- mining for diamonds, heavy minerals, gypsum, limestone, marble, monazite, kaolin, ilmenite and titanium in the Sandveld, Southern Namib Desert, Vanrhynsdorp (Knersvlakte) Centre and Richtersveld bioregions;
- illegal collection of succulents and bulbs.
- predicted effects of climate change (Rutherford *et al.* 1999)

Given its position as a global biodiversity hotspot as well as poor conservation status, the vulnerability of its biodiversity to alternative land-uses, and the potential availability of large tracts of land for reservation, a systematic approach to the conservation of the biome is long overdue. Recently, the SKEP project was initiated to address these issues (Driver *et al.* 2003b). One outcome of the project was the identification of nine geographic priority regions for immediate conservation action, of which the Knersvlakte was one.

5.1.3 The Knersvlakte

The Knersvlakte, or Vanrhynsdorp Centre, is a bioregion within the Succulent Karoo (Hilton-Taylor 1994b) (Figure 5.1). The area, comprising approximately 10 000 km^2, is home to about 133 Red Data Book plant species (Hilton-Taylor 1994b). An analysis of the SKEP QDS herbarium data puts this estimate at 127 species (Table 5.1). The region is renowned for its rich flora of minute succulents associated with quartz fields (Schmiedel and Jürgens 1999; Schmiedel 2002). Other hard rock substrata such as quartzite and limestone also support a biologically interesting and distinct flora (P. Desmet and A. Ellis, unpublished data). The intervening matrix of heuweltjie veld on reddish, colluvial sandyloams is botanically comparable, but generally lacks range-restricted endemics. Hilton-Taylor (Hilton-Taylor 1994b), Schmiedel (Schmiedel and Jürgens 1999; Schmiedel 2002) and Ellis (Ellis 1999) discuss the biophysical and biological environment of the Knersvlakte in more detail.

Table 5.1 A summary of the QDS herbarium data for the Knersvlakte Bioregion as defined by Hilton-Taylor (Hilton-Taylor 1994a). Data from PRECIS via the SKEP Project (Driver *et al.* 2003b).

Summary statistic description	Total Number
QDS with >50% area in bioregion included	14
Plant species recorded in these QDS	1389
Plant species assessed to be Succulent Karoo species	1210
Plant species with RDL classification	127

The Knersvlakte has long been recognized as a priority region for plant conservation (Hilton-Taylor 1994b; le Roux and Simpson 1994; Cowling *et al.* 1998; Lombard *et al.* 1999a). Both the provincial conservation authority (WCCB) and South African National Parks (SANP) have expressed interest in establishing a system of conservation areas in the region (Hilton-Taylor 1994b; le Roux and Simpson 1994).

Of key interest in the Knersvlakte are the quartz-patch and limestone floras and the local-scale ecological processes that have resulted in the phenomenal radiation in succulent and geophytic plant lineages on these habitats. These aspects of the regions biodiversity form the focus of the initial core reserve design discussed in this chapter. The biosphere reserve, discussed in the following chapter, focuses on all the biodiversity of the region and larger-scale ecological processes that allows biodiversity to persist over climate-change and geological times-scales.

5.1.4 A Conceptual Outline of the Conservation-Planning Approach

This section outlines the conceptual basis of the conservation planning approach used for the Knersvlakte study. The concepts and analytical tools used in this study reflect the most recent advances in systematic conservation planning.

The past 20 years have witnessed major methodological and conceptual shifts in conservation planning. The first major shift was from *ad hoc* reserve establishment to systematic protocols that identify whole sets of complementary areas which collectively achieve some overall conservation goal - the "minimum set" approach (Pressey *et al.* 1993). In this strategy, the conservation goal consists of quantitative targets for each species (e.g. at least one occurrence) or each habitat (e.g. at least 10% of its total area). The aim is to represent the required amount of each species or habitat in as small an area as possible. Usually, rapid implementation of the reserve system is assumed implicitly, so there is no basis for deciding how to schedule conservation action in relation to prevailing threats.

A more realistic scenario is for implementation of the reserve system to take years or decades, during which time the agents of biodiversity loss continue to operate. In such situations, strategies for maximizing representation on paper must be complemented or replaced by those that maximize "retention" in the face of ongoing loss or degradation of habitat. Although it is possible to do conservation planning using only the conservation value of a site, a crucial consideration for maximizing retention is the assignment of priorities based on both the conservation value of a site or irreplaceability, and its vulnerability to biodiversity loss as a result of current or impending threatening processes (Pressey *et al.* 1996). Areas of high irreplaceability and high vulnerability are the highest priorities for conservation action. This approach is intended to minimize the extent to which representation targets are compromised by ongoing loss of habitat and species.

The most recent conceptual shifts in conservation planning address the long-term persistence of biodiversity. The implementation of reserve systems that are designed to achieve only the representation of biodiversity pattern will not ensure long-term conservation. This is because these systems do not explicitly consider the ecological and evolutionary processes that maintain and generate biodiversity (Cowling *et al.* 1999a). The ultimate goal of planning should be the design of conservation systems that enable biodiversity to persist in the face of natural and human-induced change. Design is defined here as the size, shape, connectivity, orientation and juxtaposition of conservation areas intended to address issues such as viable populations, minimization of edge effects, maintenance of disturbance regimes and movement patterns, continuation of evolutionary processes, and resilience to climate change.

It is important to note that "design of conservation systems" refers to the whole landscape and not simply strict conservation areas. It is unrealistic to expect that formal reserves will be sufficient to conserve both biodiversity patterns and processes. The previous Knersvlakte study (Desmet *et al.* 1999) and the CAPE project (Cowling *et al.* 2003b) illustrate how "land hungry" ecological processes are if the goal is to include these large-scale processes within formal reserve. In this project, systematic conservation planning is applied at two spatial scales: the local reserve design scale and the broader whole landscape scale. The local scale addresses representation issues within a core reserve that targets the unique biodiversity of the region. The broader-scale biosphere reserve targets landscape-scale ecological processes that need to be conserved but which cannot realistically be conserved within a single statutory reserve.

Given that the implementation of reserves systems is almost always gradual, and accompanied by ongoing loss of habitat, the conservation of both pattern and process will require consideration of:

- Representation of biodiversity and reserve design principles in the identification of potential conservation areas; and
- Sound decisions about scheduling implementation of conservation action so that alternative land-uses have minimal impact on the desired outcome.

Systematic conservation planning is therefore about promoting both retention and persistence. Retention maximises biodiversity maintained in the face of on going loss whilst persistence explicitly incorporates ecological processes. In the implementation phase of a conservation system, incorporating information into the planning process on competing land-uses that could compromise the achievement of representation and design goals is key to maximising retention (Cowling *et al.* 1999a). This strategy achieves greater long-term benefits for biodiversity than alternative strategies based only on the representation of biodiversity pattern.

Given this conceptual framework for conservation planning, an explicit and logical protocol for reserve design in the Knersvlakte was developed for this project (Table 5.2). The protocol, based on that of Cowling *et al.* (1999a) and Margules and Pressey (2000a), comprises a series of steps that are required to identify and begin implementing a conservation system designed for the persistence of biodiversity. A key focus of this approach is the retention of both ecological pattern (e.g. representation of species or habitats) and process (maintenance of demographic processes or migratory pathways). The steps involve the design and identification of initial conservation priorities for both the core and biosphere reserve. The conceptual outline for the biosphere reserve considered in step 7 is introduced in the following chapter.

Table 5.2: Steps in the conservation planning protocol used in this study. Adapted from Cowling *et al.* (1999a) and Margules and Pressey (2000a).

Steps	Action
Step 1	Compile data on the biodiversity of the planning region and identify natural features (e.g. species, habitats, as well as spatial components of the region that act as surrogates for ecological and evolutionary processes) to be targeted by conservation action.
Step 2	Identify conservation goals for the planning region including targets for biodiversity features.
Step 3	Review the efficacy of the existing reserve network.
Step 4	Identify alternative land-use options that could potentially compromise the achievement of the conservation targets.
Step 5	Layout options and design a core conservation area that achieves the representation targets for a focused subset of key biodiversity features.
Step 6	Assign priority for conservation action for the core reserve based on the potential for selected areas to be lost to competing land-uses.
Step 7	Design an extended conservation framework for the biosphere reserve that achieves all representation, ecological process and landscape functionality targets that compliments the core reserve.

Implementation issues are not discussed in detail here, as the focus of this study is the design of a conservation system. An introduction to implementation issues in South Africa can be found in Driver *et al.* (2003a). This study differs from other recent conservation planning studies South Africa in that it is a demand driven plan. The core reserve is linked to a WWF and WCNCB initiative discussed in the introduction to this chapter. The biosphere reserve, although requested by the WCNCB for this study, is linked to a provincial bioregional planning initiative. This is discusses in more detail in the following chapter.

5.1.5 Definition of the Planning Domain

The planning domain for this study covers those parts of the Western Cape Province that fall into the Matzikamma Municipality and the West Coast District Management Area (Figure 5.1). This area includes the majority of the Knersvlakte bioregion (Hilton-Taylor 1996)(Figure 5.1) and surrounding coastal plain, lower Olifants River and escarpment environments. The planning domain is delimited along political boundaries to facilitate the integration of the project outputs into regional and local land-use planning.

5.1.6 Software

Spatial analyses in this project were performed using ArcView geographic information system software with the Spatial Analyst and 3D Analyst extensions. Microsoft Excel and Access were used to manipulate databases for use in ArcView and C-Plan. This document was produced with Microsoft Word. Graphics were generated using ArcView or Microsoft PowerPoint, and manipulated in Adobe Photoshop 7 and IrfanView.

The C-Plan program is a software package developed by the New South Wales Parks and Wildlife Service as a conservation planning decision support tool (Anon. 2001). This tool was developed to assist conservation and land-use planners to identify and evaluate spatial options and trade-offs for the development of conservation systems. It is a stand-alone program with and extension add-in for ArcView (ESRI, Redlands, California).

The program prioritizes parcels of land or sites (e.g. cadastres) based on a computed measure of conservation value, namely irreplaceability (Ferrier *et al.* 2000). The irreplaceability index is a measure assigned to a planning unit that reflects the importance of that site, in the context of the planning domain, for achieving conservation targets for a given set of biological features. Features can be vegetation types, habitats, species or spatial surrogates for processes.

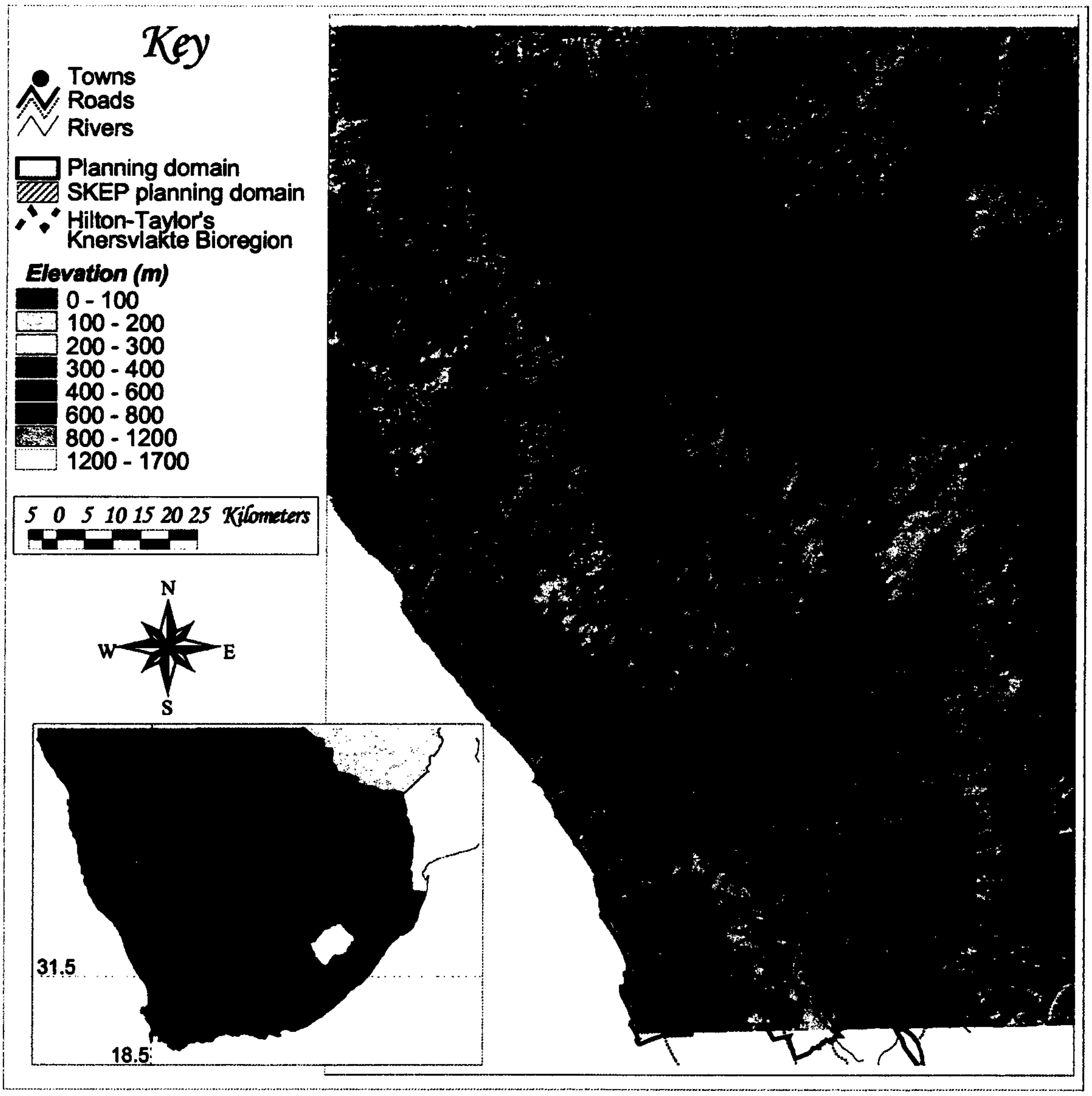


Figure 5.1 The boundaries of the planning domain and the proposed Knersvlakte Biosphere Reserve that incorporate the Matzikamma Municipality and the West Coast District Management Area.

Site irreplaceability is a function of how much of each target is achieved. Thus irreplaceability can be viewed in two ways (Margules and Pressey 2000a; Anon. 2001):

- The potential contribution of any site to a conservation goal or the likelihood of that site being required to achieve the goal.
- The extent to which the options for achieving a system of conservation areas, which is representative (i.e. achieves all the targets), are reduced if that site is lost or made unavailable.

As land is "reserved", C-Plan updates the irreplaceability index for each unreserved site to reflect how much that site contributes towards achieving the remaining conservation target. Sites with a high irreplaceability value are essential components of the reserve system if targets are to be met (i.e. if that site is not included in the reserve system then it is unlikely that targets will be achieved). Low site irreplaceability means that there is flexibility in terms of which sites can be chosen to achieve the target.

C-Plan does not provide explicit solutions for conservation area systems. It does, however, enable the evaluation of informed conservation decisions in terms of irreplaceability. After each decision, the irreplaceability of each remaining available site in the planning domain is recalculated and displayed on screen. Therefore, it is possible to objectively compare the tradeoffs between different reserve designs by comparing how each configuration contributes towards achieving a set of targets.

5.2 Step 1: Compile data on the biodiversity of the planning region and identify natural features to be targeted by conservation action.

This is the first step in the systematic planning protocol. It involves the identification of the spatial components of biodiversity that need to be considered in the conservation system. Some of these will be elements of biodiversity pattern, e.g. a key species or a particular habitat. Others will serve as surrogates for the ecological and evolutionary processes that should be maintained in the landscape, e.g. upland-lowland gradients. Essentially the question being asked here is "what data are available and appropriate as biodiversity surrogates to target for this plan?" All the biodiversity surrogates identified are collectively termed "biodiversity features" in this planning context.

Biodiversity is a continuous entity across spatial and temporal scales. The goal of the biodiversity feature data used in planning is to act as a convenient surrogate that captures as much of the variation observed in this biodiversity (Margules and Pressey 2000a). Any biodiversity surrogate data, such as species, populations, habitats or vegetation types, can be used provided that they adequately reflect this goal, and are spatially explicit.

The planning process involves making decisions about pieces of land based on the information attached to those pieces of land. Thus, all data used needs to be spatial. In other words each feature needs to be linked to some point, line or area on a map of the area of interest in order for it to be useful. This fundamental property of the process applies to all data types used, not only biodiversity data.

In addition, as the planning process makes decisions about all areas in the planning domain, at least one of the biodiversity feature layers need to be continuous, such as a vegetation map. The absence of such a data layer will mean that some areas will effectively be ignored in the planning process, as they have no biodiversity information attached to them.

In addition to feature layers representing biodiversity pattern (i.e. the distribution of biodiversity in the landscape), it is important to include feature layers on biodiversity processes in the planning process (Cowling *et al.* 1999a; Margules and Pressey 2000a). Biodiversity is not a static entity. Individuals, species, populations and even habitats interact with one another and their abiotic environment and are able to persist and evolve in response to changing conditions. These complex webs of interactions are collectively termed ecological processes. Explicit consideration and inclusion of processes in developing a regional conservation plan is a key step toward designing what can be termed a "living landscape". A landscape that continues to function naturally, allowing species to persist, migrate and evolve in the context of continued human utilisation and development, and a dynamic natural environment.

In addition to the biodiversity information layers identified, it is possible to include information layers of non-biodiversity related features such as landscape aesthetics or cultural heritage features. This information is especially important in the context of the properties of a biosphere reserve. Cultural features are also an integral component of biosphere reserves. The viewshed analysis discussed in Section 5.2.3 attempts to incorporate one aspect of societies valuation of conservation areas. The concept of a protected viewshed is incorporated into the design of the biosphere reserve (see Section 5.2.3).

For ease of discussion the biodiversity features, datasets and planning units developed for the core reserve and biosphere reserve are both discussed in this chapter.

5.2.1 Biodiversity Pattern Data

Four groups of biodiversity information were used in this study. All comprise information layers of continuously mapped higher-order biodiversity features such as a vegetation types or habitats. Finer-scale and lower-order biodiversity information such as point species or population distribution records were not used as this type of data is generally not available for the entire planning domain.

Each biodiversity pattern layer was overlaid on the SKEP transformation layer to estimate the remaining extent of each biodiversity feature. This transformation layer was developed as part of the SKEP process (Driver *et al.* 2003b). It is based primarily on the 1996 National land-cover as well as expert input, mapped transformation and degradation by the Department of Agriculture and the early-1990's NASA landsat-5 false-colour mosaic of southern Africa. Transformation is defined here as any area that has been irreversibly converted from a natural or near-natural vegetation state such as agricultural fields and urban areas. This information layer is at best a rough approximation of the extent of transformation. For most areas this will be an underestimate of the present state of transformation. Understanding patterns of transformation in the landscape are key to understanding priorities for conservation, as patches of natural habitat embedded in highly transformed areas tend also to be priority areas for conservation (Pressey and Tully 1994; Pressey and Taffs 2001a; Cowling *et al.* 2003b) (see also Chapter 6.5).

Four biodiversity pattern features are used in this study. These are the SKEP vegetation map, a modelled land-class map, a map of key habitats and the SKEP expert-mapped areas of biological importance.

5.2.1.1 SKEP vegetation map

The SKEP vegetation map was used as the primary biodiversity pattern feature information layer for planning (Figure 5.2). A total of 35 vegetation types fall within the planning domain. This vegetation map was derived from the new South African vegetation map being prepared by the National Botanical Institute. A full description of how the map was developed is contained within the SKEP Technical Report (Driver *et al.* 2003b). The advantage of using this vegetation map is that it covers the entire Succulent Karoo, thus allowing determination of each vegetation types' global extent. Also, it is possible to determine which vegetation types have their core distribution in the planning domain and which occur only peripherally.

Unfortunately, the SKEP vegetation map does not cover the entire planning domain. In the extreme south of the planning domain there are areas that are not covered by this map (Figure 5.2). This does have implications for planning, as irreplaceability values for these areas are not accurate relative to other sites in the planning domain. This omission is not viewed as critical as these areas are peripheral to the core areas of concern in this study.

5.2.1.2 Land-class map

A land-class map was used as second continuous biodiversity pattern feature information layer (Figure 5.3). A total of 78 land-classes fall within the planning domain. The land-classes act as another type of biodiversity surrogate similar to vegetation types. This map was developed for Namaqualand in conjunction with Simon Ferrier and Glen Manion from the New South Wales Parks and Wildlife Service in Armidale, Australia. The method used generalised additive modelling to combine indices of soil and climate, and then to classify all possible combinations into biologically meaningful classes based on a cluster analysis of vegetation community data that was compiled for the region. All input environmental data layers were in ArcInfo grids at a 100m grid-cell size resolution. The vegetation community database comprised a collection of 5567 phytosociological relevé samples from across Namaqualand (Appendix 5.4). It must be remembered that the accuracy of a modelled land-class map is only as good as the environmental data used to develop it. Often this data is modelled to begin with, e.g. extrapolated rainfall data, and so errors in the extrapolated environmental data are compounded in the final land-class map. No error estimation has yet been performed on this land-class map.

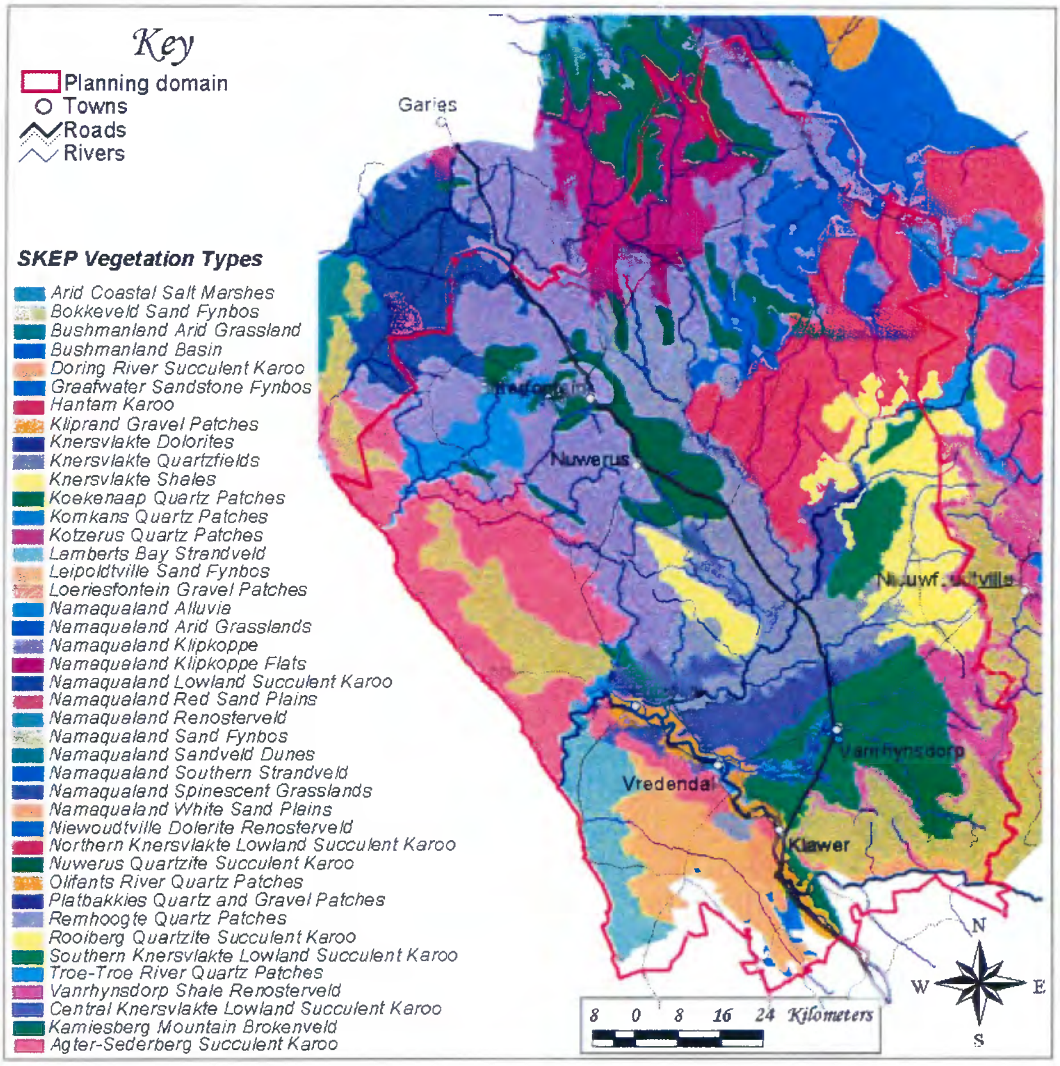


Figure 5.2: SKEP vegetation types that occur in the planning domain.

The advantage of incorporating the land-class map is that it adds "resolution" to the SKEP vegetation types. The land-class map is in many cases able to subdivide large vegetation type polygons into finer-scale units. Planning units that would otherwise be considered equal should they all occur in the same vegetation type can now be differentiated by adding the land-class information. The utility of land-class data in conservation planning is discussed in more detail in Chapter 2

The same problem afflicting the vegetation map of missing data befalls the extreme south of the planning domain for the land-class map.

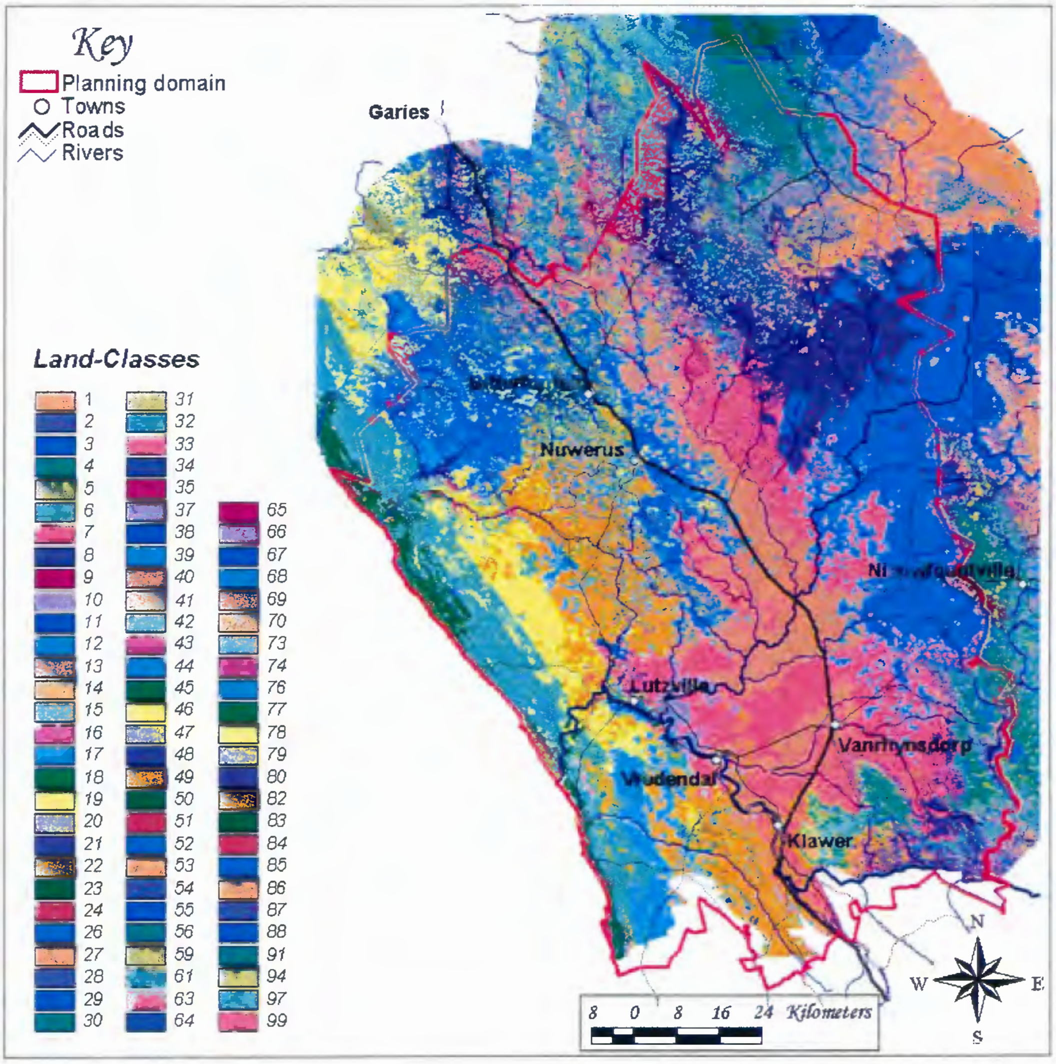


Figure 5.3: The land-class map for the planning domain. 78 land-classes fall within the planning domain.

5.2.1.3 Focus habitat type map

The Knersvlakte is renowned for its quartz-patch and limestone habitats. These two habitats, plus quartzite rock habitats, are home to the majority of endemic species in the Succulent Karoo areas of the planning domain (Desmet *et al.* 1999; Schmiedel and

Jürgens 1999). Together these three habitats comprise the "focus" habitat types biodiversity feature pattern layer identified here for inclusion in a statutory reserve in the region (Figure 5.4; Table 5.3). For the original Knersvlakte study (Desmet *et al.* 1999) these habitats were mapped from a combination of satellite imagery and 1:10 000 scale orthophotographs. These habitats were sub-divided into geographic regions (e.g. north-west or south-east) to reflect the turnover of species as one moves from one area of quartz-patches to the next in the planning domain. Unlike the previous two feature layers, this is a non-continuous biodiversity information layer. These habitats were not mapped as discrete units in either the vegetation or land-class map. On these maps these habitats fall within more broadly mapped (1:250 000 scale) quartz patch vegetation types or land-classes. Thus, some small outlying areas of quartz patches fall within different vegetation or land-class types.

Again there is a problem with missing data in this information layer but not due to the coverage having been truncated. For the original Knersvlakte study the focus of attention was on the quartz patches in the centre of the planning domain. The expansion of the planning domain in this study to the boundaries of the Western Cape Province north of the Olifants River includes another large area of quartz patches along the lower Sout River north of Brandsebaai (Note that this is a second Sout River and is not be confused with the Sout River that runs through the middle of the planning domain). This area of quartz patches is not reflected in focus habitat types map (Figure 5.4).

Table 5.3: A summary of the focus habitat types and their aerial extent within the planning domain. This spatial extent represents the global distribtuion of these habitats as well as the vegetation that occurs on them.

Focus habitat type name	Original extent (ha)	Estimated area remaining (ha)	Estimated % Transformed
Western quartz-patches	2263.25	2069.75	8.55
Western intermediate heuweltjie/quartz veld	1252.50	1218.25	2.73
Northern quartz-patches	11383.50	11203.25	1.58
Northern intermediate heuweltjie/quartz veld	5248.25	5244.75	0.07
Central quartz-patches	13130.00	13105.50	0.19
Central intermediate heuweltjie/quartz veld	5345.75	5345.75	0
Southeast quartz-patches	1267.75	1000.00	21.12
Southeast intermediate heuweltjie/quartz veld	757.25	681.50	10
Southwest quartz-patches	421.75	190.00	54.95
Southwest intermediate heuweltjie/quartz veld	32.50	3.00	90.77
Limestone	4920.75	4776.00	2.94

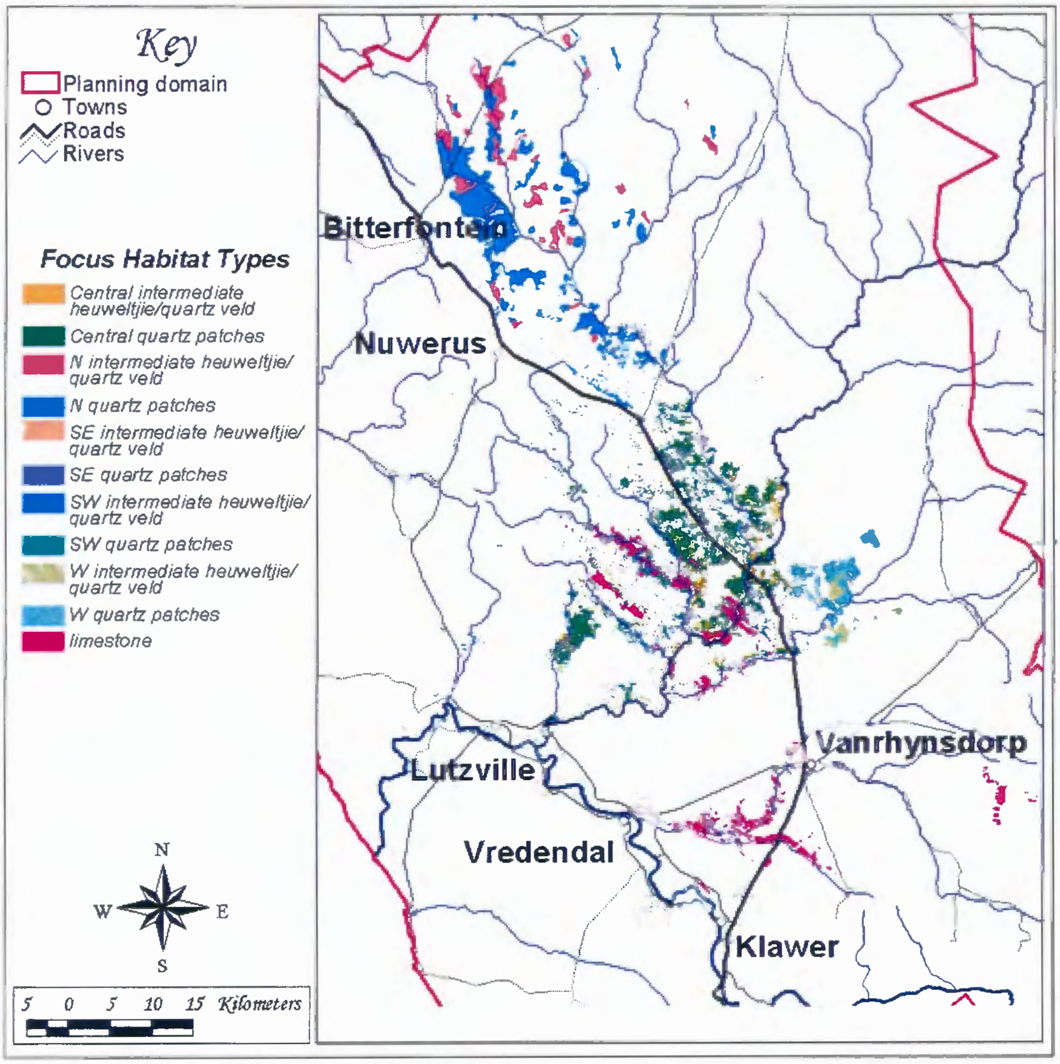


Figure 5.4: The distribution of focus habitat types within the core area of the planning domain. Note that the quartz patches in the north west of the planning domain along the lower reaches of the Sout River (number 2) north of Brandsebaai are not indicated on this map.

5.2.1.4 SKEP expert-mapped areas

The SKEP expert-mapped areas were used as a fourth biological pattern information layer (Figure 5.5). This map, developed as part of the SKEP process, used experts on plants, amphibians, fish, birds, invertebrates, mammals and reptiles to map on 1:250 000 maps areas that they consider to be of biological importance in the Succulent Karoo (e.g. local centres of diversity, key habitats, etc.). Details on the expert mapping process are contained in the SKEP Technical Report (Driver *et al.* 2003b).

The inclusion of the expert information is an attempt to integrate the wealth of knowledge that the respective researchers have accumulated through years of work in the Succulent Karoo, but which is not necessarily contained in formal publications or reflected in any of the other biodiversity pattern or process information layers. This information is similar to museum or herbarium collections data in that it is presence only data. Experts can map only that information that they know and not what they don't know. The fact that an expert does not map an area does not imply that it is not important in the broader scheme of things. The absence of expert areas in the lower reaches of the Sout River (number 2) north of Brandsebaai or the sand-plain fynbos in the Haartebeesekom are cases in point implying that there are unique biodiversity features there that were unknown to the experts at that time when the SKEP maps was made.

5.2.2 Biodiversity Process Data

As with biodiversity pattern data, ecological processes need to be represented spatially if they are to be incorporated into the planning process. Appendix 1 presents a preliminary synthesis of the processes required for maintaining and generating plant biodiversity on the Knersvlakte, together with the spatial components that sustain them and the temporal scales over which they operate. It is a formidable list but by no means a comprehensive one. Naturally, to represent all these processes spatially in the planning process will be impractical if not impossible. Consequently just as a few meaningful surrogates are used to represent biodiversity pattern, so processes need to be represented by a tractable number of process surrogates that can be mapped spatially.

Many processes that maintain biodiversity act over medium to small spatial scales less than 10 000 ha. Selecting one or two planning units for other biodiversity features will by default conserve these processes. Other processes happen over long temporal scales of thousand to millions of years, and these are beyond the scope of our immediate consideration. Processes that occur over the medium to large spatial scales (>10 000 ha to landscape scale) and short to medium term (<100 years) are targeted in this project. These are processes that cover a few or several dozen planning units and are likely to be observed operating within our lifetime. Conserving these processes requires explicit consideration in regional plans as they tend to cover numerous planning units and target a large proportion of the planning domain. The scale of focus needs to be calibrated to the resolution of the planning project. For example, in contrast to the current approach if planning units were 1 ha grid cells and the entire planning domain less than 1000 ha, then one would focus on individual- or population-level processes rather than landscape-level processes.

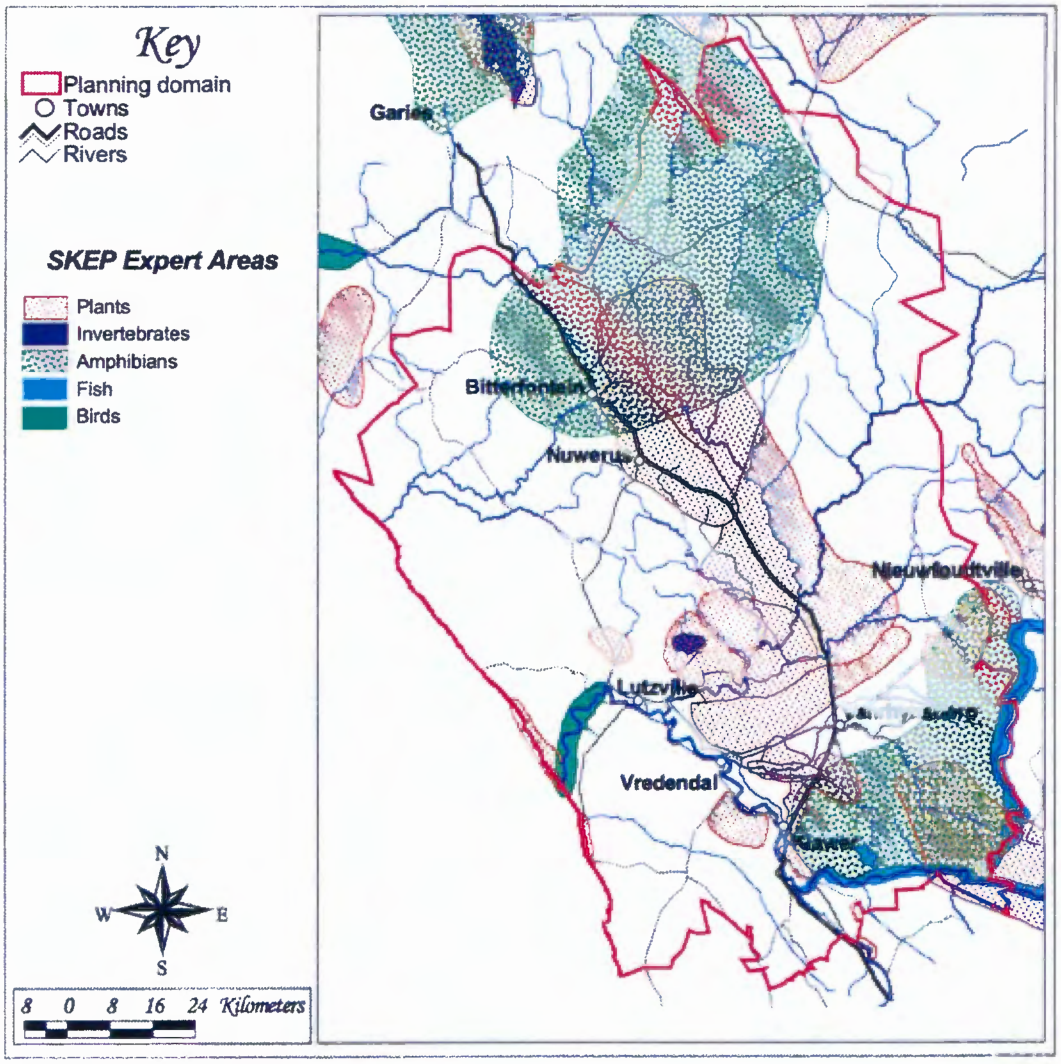


Figure 5.5: SKEP expert map of known areas of biodiversity importance in the planning domain.

Processes are considered at two scales in this project. For the core reserve the focus is on representing biodiversity pattern and those small to medium-scale processes that would operate within the confines of a single reserve. Thus, for the core reserve planning considered in this chapter processes are not explicitly represented as features in the planning dataset. The focus of the biosphere reserve is at the landscape scale and the larger-scale processes talked about above are explicitly incorporated as features in the biosphere reserve planning dataset.

The processes used in this project incorporate applications from other projects (edaphic interfaces and riparian zones were used in CAPE, STEP and SKEP) and also novel interpretations of processes derived specifically for this project (topographic climate refugia). All these processes operate at the landscape-level.

In addition to the three groups of processes "hard-wired" into the planning feature dataset, upland-lowland and biogeographic gradients were also considered in planning through the use of reserve design rules (see Section 5.3.3). See Section 6.3 for a discussion on the rationale for using both "hard-wired" and "non-hard-wired" processes features in the planning process. Upland-lowland and biogeographic gradients over short spatial scales are important for promoting persistence in the face of climate change and capturing evolutionary gradients respectively (Desmet *et al.* 1999; Cowling and Pressey 2003; Rouget *et al.* 2003a)

5.2.2.1 Edaphic interfaces

Major edaphic discontinuities, such as those between vegetation types occurring on substrates derived from different parent material (e.g. aeolian sand vs. colluvial loam), are important for evolutionary processes (Cowling *et al.* 1999a). For plant species to migrate across these interfaces requires the evolution of novel traits to be able to adapt to the conditions of the new substrate. Habitat specialisation, such as this, plays an important role in plant diversification in the succulent karoo (Desmet *et al.* 2002; Schmiedel 2002).

These interfaces can also be regarded as ecotones between two functionally different substrates. As such they are usually areas of higher species diversity (Spector 2002), partly because there are more habitats present. These also represent the extremes of species ranges where the selective pressure of the neighbouring habitat can result in these populations being genetically distinct from those in the core distribution of the species (Heywood 1986; Turner 1989; Montana *et al.* 1990; Holland *et al.* 1991; Gosz 1993; Margalef 1994; Kent *et al.* 1997).

Boundaries between vegetation types on similar soils can be considered soft edaphic interfaces. These interfaces are important for plant species migration as they allow species to move relatively unhindered in the face of changing climates.

In the planning domain, classifying each unique combination of vegetation type on the basis of their underlying parent material was used to identify edaphic interfaces. This follows the same approach as that used in the SKEP project (Driver *et al.* 2003b). Unique combinations were coded as hard, semi or soft interfaces according to the potential for species to move across these edaphic boundaries (Table 5.4). Only hard interfaces were considered to promote ecological diversification whereas semi and soft promote migration corridors. A buffer of 500m was used on each side of all interfaces (Figure 5.6).

Table 5.4: Classification of edaphic interfaces based on unique combination of parent material and our interpretation of their potential to promote species movement (soft) or speciation (hard) (from Driver *et al.* 2003b)).

Soil Type	1	2	3	4	5	6	7	8	9
1 Acid Sand									
2 Colluvium	Hard								
3 Granite	Semi	Semi							
4 Granite & Colluvium	Semi	Soft	Soft						
5 Quartzite	Semi	Semi	Semi	Semi					
6 Quartzite & Colluvium	Semi	Soft	Semi	Semi	Soft				
7 Quartzite & Shale	Semi	Semi	Semi	Semi	Soft	Soft			
8 Sand	Semi	Semi	Semi	Semi	Semi	Semi	Semi		
9 Shale	Hard	Hard	Hard	Hard	Hard	Hard	Soft	Hard	
10 Alkali Sand	Hard	Soft	Hard	Soft	Hard	Soft	Hard	Semi	Hard

Edaphic interfaces mapped at the scale of the vegetation type do not capture all types of edaphic interfaces. Allan Ellis (unpublished data) and Schmiedel (Schmiedel and Jürgens 1999; Schmiedel 2002) have demonstrated the evolutionary importance of edaphic interfaces at a very fine scale in the quartz-patches of the Knersvlakte. These finer-scale interfaces can, however, be captured within the average size of planning unit used and therefore it is not necessary to explicitly incorporate as feature layers in the planning dataset.

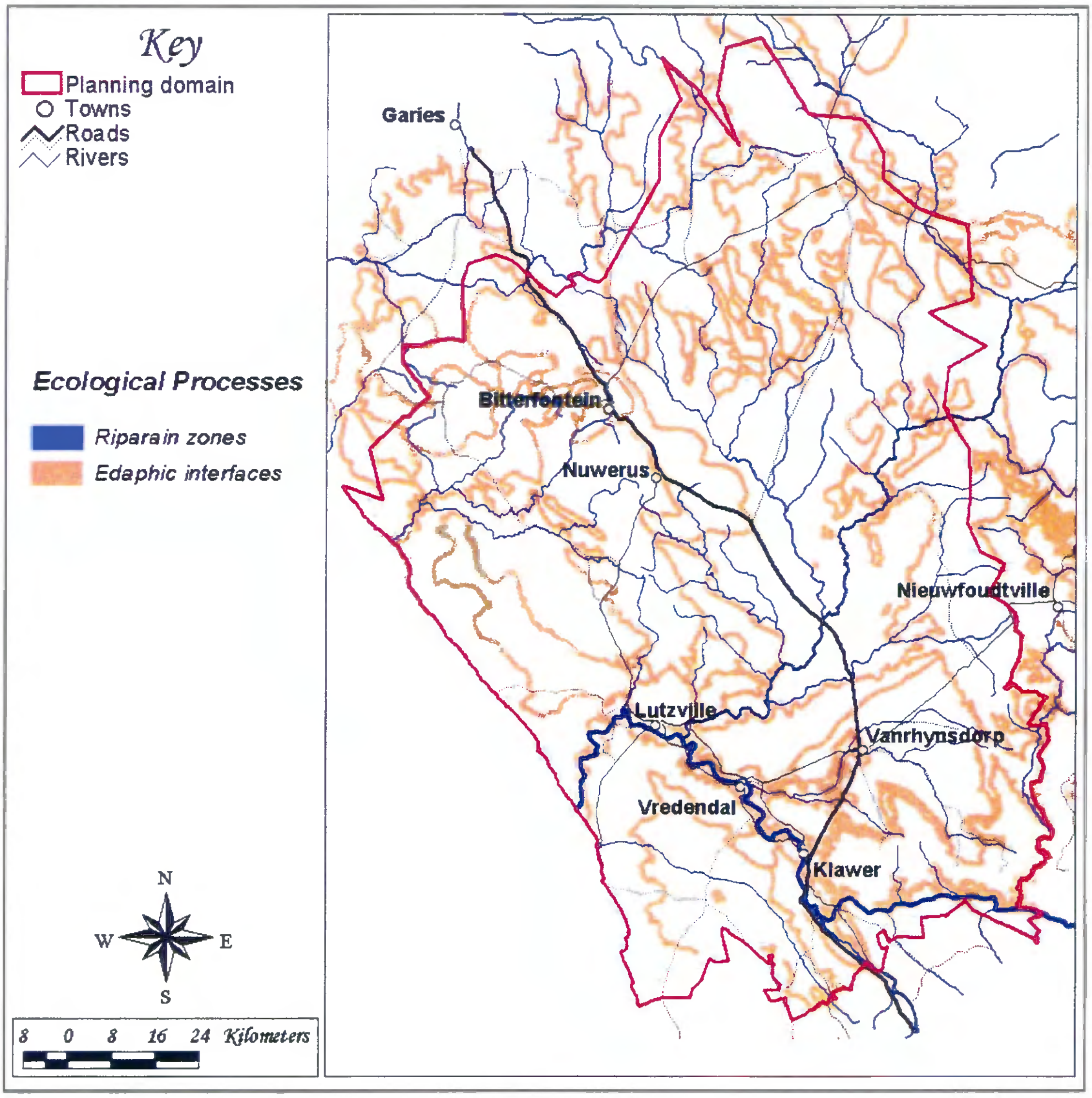


Figure 5.6: Major edaphic interfaces and river corridors mapped as components of the ecological processes.

5.2.2.2 Riparian zones

River systems are included as a surrogate for a number of key ecological processes. Rivers and riparian areas are often a key resource for fauna in the landscape as well as being corridors for faunal migration as they link different valleys and mountain systems. In addition, functional riverine habitats are important for the maintenance of hydrological processes, which have direct benefits for humans. Vegetation on river banks needs to be maintained in order for rivers themselves to remain healthy, hence the focus not just on rivers themselves but also on riparian zone.

There is evidence that mammals and birds use rivers as migration corridors (Johnsingh and Williams 1999; Meiklejohn and Hughes 1999; Gilliam and Fraser 2001; vom Hofe and Gerstmeier 2001; Robinson *et al.* 2002). There is also evidence that migration of plant species along riverine corridors has resulted in species diversification (Bayer 1999). Also, riparian zones act as refugia from drought and have provided refugia for mesic species during major climatic events in the past (Kaul *et al.* 1988, Cowling, 1999 #349). These zones often contain habitats associated with the riparian zone, for example rocky outcrops or cliffs, which are not riparian, but do provide migratory habitat stepping stones or refugia in a landscape where they are otherwise absent (Cowling *et al.* 1999a).

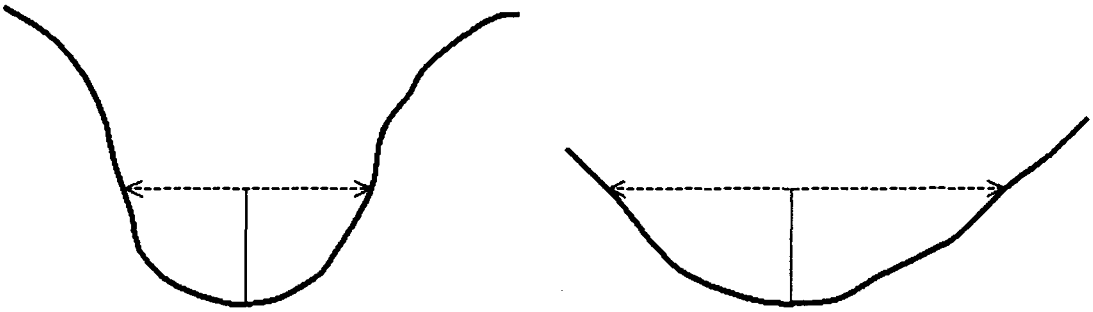
In the planning domain, a buffer was created around all rivers mapped by the Department of Water Affairs (Figure 5.6). Buffer width varied according to their stream-order categorisation of each river (Table 5.5).

Table 5.5: Department of Water Affairs stream order categories and buffer distance used in this study.

Stream Order Category	Buffer Distance (m)
1	50
2	100
3	200
4	250
5	500

A more ecologically meaningful determination of riparian buffers may be possible with the delimitation of buffers based on the width of river valleys at a given altitude above the valley floor. Figure 5.7 illustrates how a river buffer can be created using a DEM and viewshed analysis. The buffer is determined as the width of the river valley at a given altitude (e.g. 10 or 20m) above the altitude of the river or valley floor. The viewshed analysis is performed using the river course as the observation location. A different observation offset (i.e. vertical distance) can be used for different order rivers. The rationale for this method is that width of riparian corridor and associated habitats is in part determined by the width of a river valley at any given point along its course. This method was explored but not incorporated in this study.

Figure 5.7 A method for determining a more ecologically meaningful width for river buffers using a DEM and viewshed analysis. Buffers are defined based on the width of a river valley at a fixed altitude above the river valley. River valleys with steep sides (left) have narrower buffers than wide valleys (right).



5.2.2.3 Topographic climate-change refugia

Climate change is acknowledged as one of the greatest long-term threats to the persistence of succulent karoo (Rutherford *et al.* 1999; Rutherford *et al.* 2000). Abiotic features in the landscape that buffer areas in the face of a changing climate will become increasingly important conservation areas. Such features are commonly called refugia (Barthlott *et al.* 1993; Porembski *et al.* 1994; Seine *et al.* 1997; Colinvaux 1998; Danin 1999; Desmet 2000), and are characterized by having a climate that is moderated relative to surrounding landscape. As the climate changes, species are able to migrate locally into these refugia where the moderated climate allows them to persist locally (Bush 1996; Midgley *et al.* 2001). In the Succulent Karoo such refugia are typically mountain slopes or summits, especially south-facing slopes; slopes that receive the cooling influence of the coastal sea-breeze; kloofs or ravines; and, dune fields where aquifers provide more moisture than usual (Appendix 5.8). The climate moderating effect is either through higher elevation or shading leading to lower air temperatures for topographic features, or in the case of aquifers the presence of an alternative perennial water source to rainfall. Ensuring that these features remain ecologically functional and connected to the surrounding landscape will go a long way towards planning for climate change at the landscape level.

In this study three classes of topographic climate refugia are targeted (Figure 5.8). All were determined using the digital elevation model for the planning domain and ArcView's Spatial Analyst. The three landscape types were identified using the criteria presented in Table 5.6.

The area selected with each query was expanded with a buffer of 200m (i.e. two grid cells) to account for errors in the DTM and to provide a buffer for each area identified. These three landscape types are not mutually exclusive and do overlap where two or more of the criteria are met. In Table 5.6 each type represents a habitat that has a more moderated climate relative to the previous type or the surrounding landscape. For example, areas with the most moderated climates and potentially the most important refugia are those that are both south facing and under the influence of the sea breeze. Generally these refuge habitats occupy a relatively small fraction of the total planning domain (Table 5.6); however, these areas could be to most important areas for the future survival of the local biota.

Table 5.6: The area of the planning domain occupied by the three classes of topographic climate refugia identified.

	Topographic Refugia Type	Area of Planning Domain	
		(ha)	(%)
1	Slopes greater than 15°	225024	16.67
2	South facing slopes (>110° and <250°) greater than 15°	98164	7.27
3	Slopes greater than 15° that are in line-of-sight of the ocean breeze	62835	4.65

5.2.3 Landscape Aesthetics

A key component of the core Knersvlakte reserve will be the preservation of views both within and surrounding the park. Visitors to a park want to experience nature and natural landscapes, and landscapes broken with human infrastructure detracts from this experience. This becomes particularly important in a relatively flat landscape like the Knersvlakte. A visibility surface or viewshed for the core reserve was created of the surrounding landscape from a number of observation points located throughout the proposed reserve (Figure 5.9). This viewshed indicates all areas in the planning domain that can be seen from within the park. Although the viewshed is not strictly a surrogate for ecological processes it is a landscape-scale feature that covers a large proportion of the planning domain. The viewshed is not a biodiversity feature but rather an aesthetic feature that relates to societal values placed on the core reserve. As the viewshed is "land hungry" in its requirements including it in the biosphere reserve design can help in the location of buffer and corridor areas that fulfil both biodiversity criteria as well as social criteria.

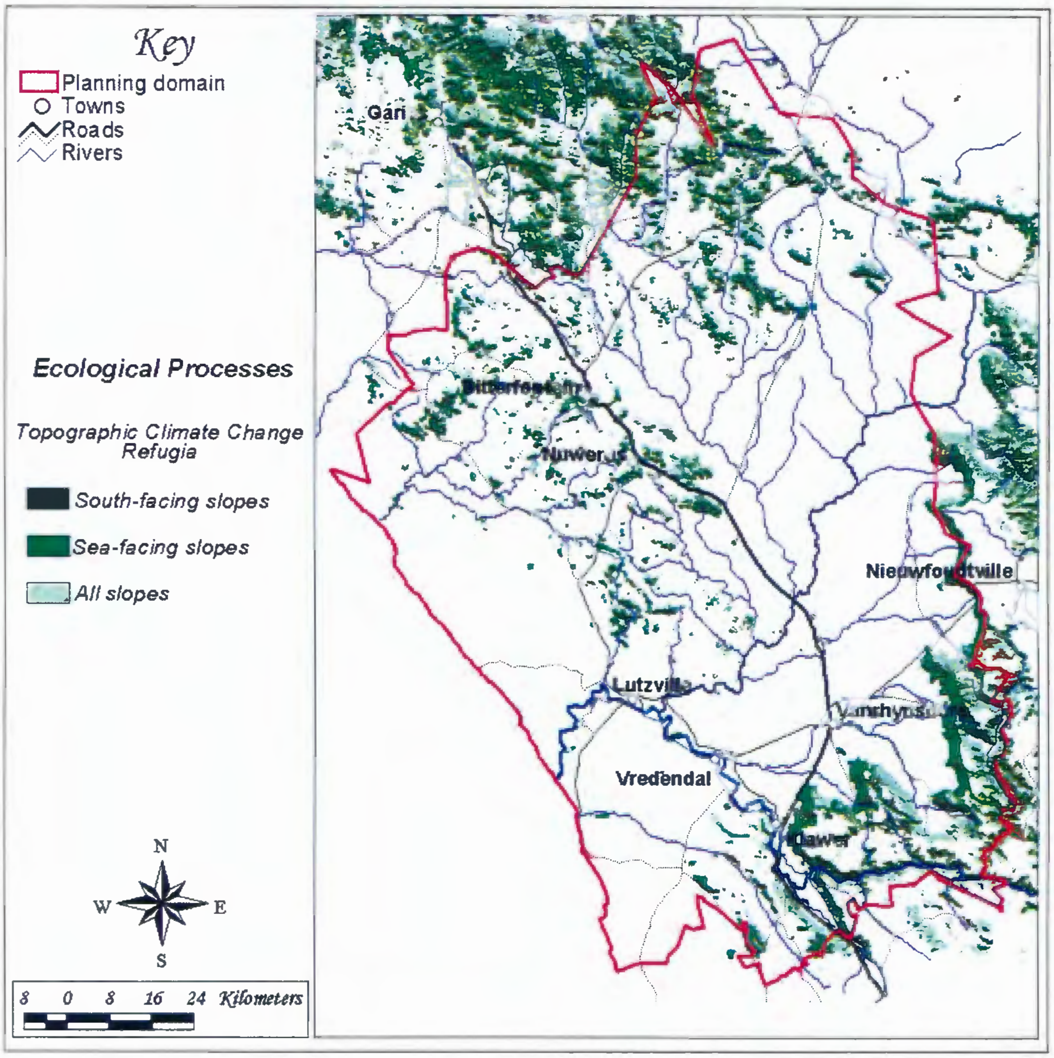


Figure 5.8: topographic climate refugia associated with mountainous area in teh planning domain.

Development within line of site of the proposed park needs to be sensitive to the needs of the park. For example, transformation of large tracts of land for agriculture or a new mine within the park viewshed will detract from the development potential of the park. Land-use planning viewshed protection and management can be used in preserving scenic urban and natural landscape (Nagy 1994; Bacon 1995; Fisher 1996; Camp *et al.* 1997) (See also http://www.sustainable.doe.gov/, http://www.dot.ca.gov/, http://www.co.napa.ca.us/departments/planning).

The viewshed was calculated using 3D Analyst in ArcView from the DEM for the planning domain and 14 observations points located within the proposed core reserve (Figure 5.9).

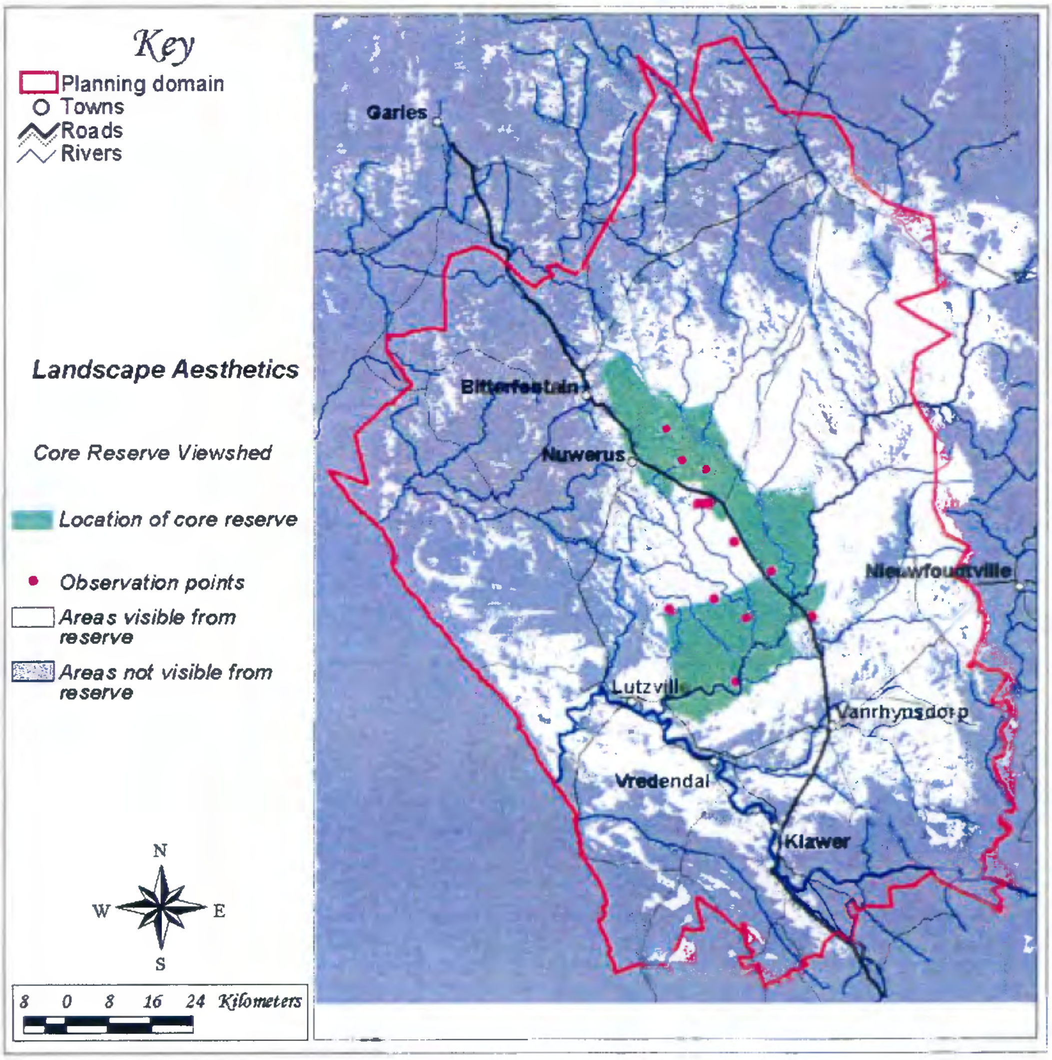


Figure 5.9: The viewshed of the of the proposed core Knersvlakte reserve.

5.2.4 Contextual Spatial Data

In addition to the biodiversity feature information discussed above, several non-biodiversity related information layers were used either as contextual information; as source data for derivation of the biodiversity feature layers; or, in the prioritisation of areas for conservation action (Table 5.7).

Table 5.7: A summary of non-biodiversity related spatial data layers used in this project.

Information Description	Original Source of Data	Data Provider for this Project
Digital elevation model at 100m grid cell-size resolution	Computa Maps, Cape Town	Philip Desmet
Rivers	Department of Water Affairs	CAPE Project
Roads	Department of Water Affairs	CAPE Project
Agricultural cadastral boundaries with deeds information attached	Department of Surveys and Mapping, Cape Town and Bloemfontein; Deeds Office, Pretoria	Philip Desmet
SKEP vegetation transformation map	SKEP Project	SKEP Project

5.2.5 Planning Units

Planning units are the area of land about which a decision is made in the context of the planning exercise. These units need to be relevant to both the scale of plan implementation and accuracy of the data used. There is no point using planning units smaller than the accuracy of spatial information used. For example, if the mapped vegetation boundary accuracy is approximately 250m then there is little point using planning units such as a 500m grid as there is a high probability that any boundary falling in any given planning unit may actually lie in an adjoining unit. It would be better to use a 1km grid. Likewise, if the objective of the planning exercise were to make recommendations regarding the purchase of land for conservation, such as here, it makes sense to use cadastral land parcels rather than a grid as planning units. In highly transformed landscape it may be necessary to combine cadastres with patches of remaining habitats to develop the planning unit layer. Large properties with very small proportions of critical habitats may distort priorities, as only a small proportion of these properties are required to meet conservation goals (e.g. Lombard *et al.* 1997b).

In this project two types of planning units are used based on the different objectives of the two parts of this study (Figure 5.10).

5.2.5.1 Core reserve planning units

Cadastres were used as planning units for the core reserve (Figure 5.10). Reserve development initially involves the purchase or contract of land parcels in an area into an identified reserve network. Thus, cadastres are essentially the units of plan implementation.

There are a total of 2348 cadastre-based planning units in the planning domain. Only rural cadastres are considered. Urban cadastres were grouped into town polygons. For the planning process, the SKEP vegetation types, land-classes and focus habitat types were intersected with this planning layer and a summary planning unit or site (2348) by feature extent (124) matrix generated for use in C-Plan.

5.2.5.2 Biosphere Reserve Panning Units

Grid-based units were used for planning the biosphere reserve (Figure 5.10). The goal of the biosphere reserve was to assign different land-uses to areas in the landscape based on underlying biodiversity properties. As one cadastre can fall into more than one land-use category, cadastre-based planning units are impractical for this purpose. Also, the goal of this part of the project is to identify a landscape-wide vision for land-use and and not specific properties for plan implementation.

There are a total of 13 556 1x1 km planning units. Computational efficiency was the primary determinant of the planning unit dimensions. Given the average accuracy of the data, a unit of 500x500m would have been desirable. This, however, would increase the number of units four times and significantly reduce computational speed. For the planning process, the SKEP vegetation types, land-classes, focus habitat types, SKEP expert area, edaphic interfaces, climate refugia, riparian zones and the core reserve viewshed were intersected with the transformation layer and the remaining extent of features intersected with this planning layer and a summary site (13 557) by feature extent (157) matrix generated for use in C-Plan.

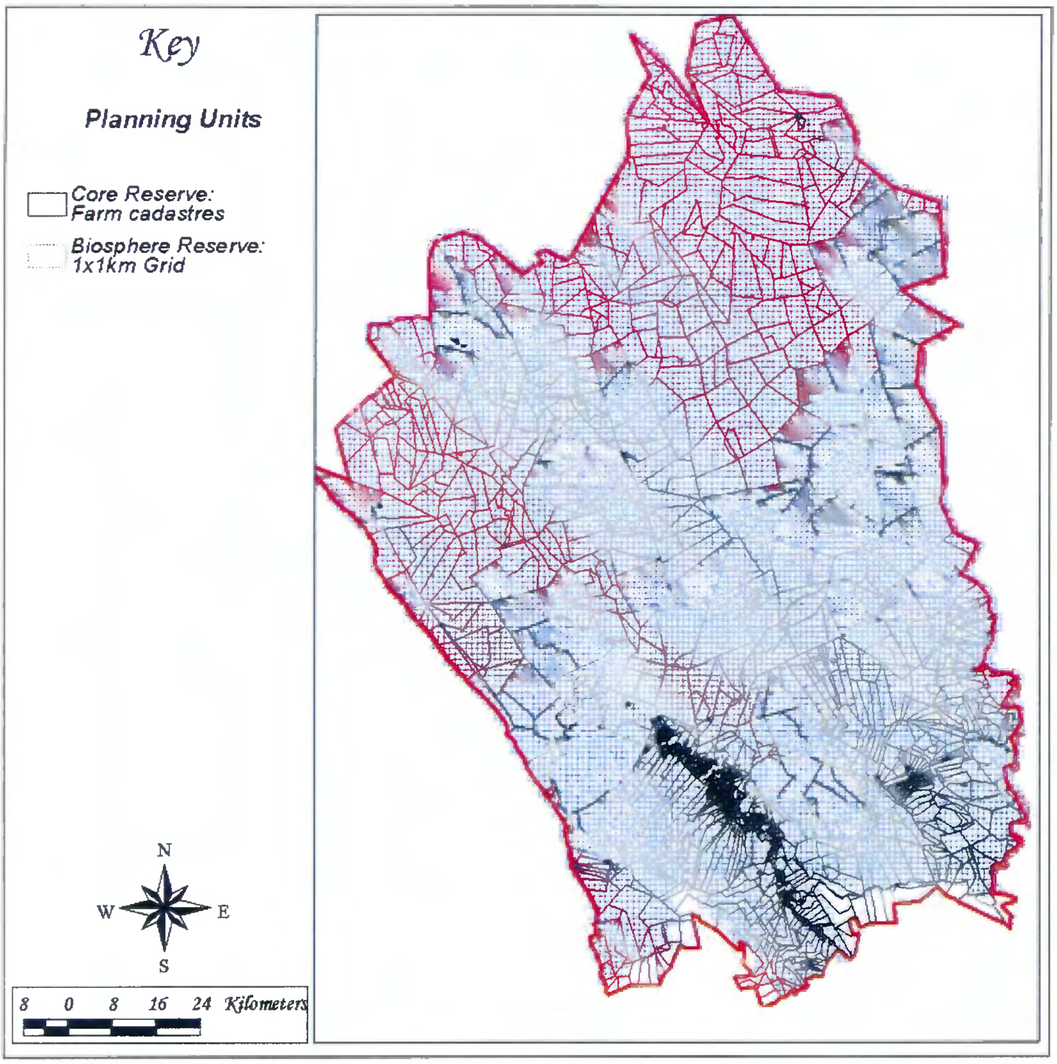


Figure 5.10: Planning units used for the design of the core and biosphere reserves.

5.3 Step 2: Setting targets for biodiversity features.

This section deals with the second step of the planning process – setting quantitative targets for the representation of the identified spatial components of biodiversity. This presents a serious challenge to conservation planners. For example, how much land is required to represent the endemic species of the planning domain? Which climatic gradients and associated juxtaposed landscapes are most likely to facilitate migration of poorly dispersed organisms in response to climate change? Setting justifiable targets for features underpins the entire systematic conservation planning approach.

"If one wishes to conserve all biological diversity, one requires the whole landscape" (Pressey *et al.* 2003a). In conservation planning a trade-off is made between the short- to medium-term needs of humans, and the need to conserve as much biological diversity as possible. Systematic conservation planning involves setting explicit targets for the conservation of biodiversity patterns and processes in order to help measure this trade-off. For example, a target might be 500 ha of a particular vegetation type, or a defined area of a particular riverine corridor. The success of the systematic approach relies on setting conservation targets in a consistent and transparent manner. Targets underpin the effectiveness of subsequent stages in the planning process.

Targets need to use the best available ecological information to interpret the conservation goals as explicit, quantitative targets for biological features. Naturally, interpretation of the goals is constrained by the availability of both quantitative and expert biodiversity information. Also, targets require periodic revision as better information comes to light. To be effective, conservation targets must meet the following three criteria:

- they must be comprehensive, i.e. they must cover all identified biodiversity features;
- they must be quantitative;
- they must be adequate and must not be constrained downwards by lack of quantitative or expert knowledge.

Targets can be divided into representation or pattern targets, and ecosystem or process targets. Pattern targets are aimed at setting aside the minimum amount of land required just to represent the biodiversity that occurs there (Pressey *et al.* 2003a). Landscapes are generally composed of repeating units of the same biodiversity features (e.g. patches of the same vegetation type or populations of the same species). Thus, conserving one, five or ten occurrences of a vegetation type might suffice for achieving the pattern target. However, for this biodiversity to persist through time requires that the processes responsible for maintaining a patch of vegetation or population are maintained. Thus, a process target is required over and above the pattern target. This could also be referred to as an ecosystem or landscape target as the goal with this target is to maintain ecosystem functioning.

A third type of target, a retention target, was used in CAPE to take account of the expected rate of future anthropogenic habitat transformation (Pressey *et al.* 2003a). This target concept is not used here. Targets should be strict products of biological criteria. Concerns about the present extent or rate of anthropogenic transformation of any feature are incorporated in the priority setting stage of the planning process. Using feature transformation information twice in the planning process would overly weight features that are at risk of being lost. In addition, this duplication would lead to confusion as to which component of a feature (viz. irreplaceability due to target versus vulnerability due to transformation) contributes more to a planning units conservation action status in the priority setting stage of planning.

The relationship between pattern and process targets is best illustrated by means of a stylised representation of a landscape (Figure 5.11). The area required to represent biodiversity pattern is the sum of the area required to represent each mutually exclusive biological feature (e.g. vegetation types or species) in the landscape, likewise for ecological processes. The relationship between these areas and different categories of land-use and the biosphere reserve is also illustrated in Figure 5.11.

Figure 5.11 can imply that statutory conservation areas be identified to conserve biodiversity pattern. As pattern is the foremost aspect of biodiversity to be lost in the face of transformation as well as being the building blocks of biodiversity it is a sensible approach for statutory conservation mechanisms to focus primarily on ensuring the retention of pattern. Naturally, small to medium temporal and spatial scale processes also need to be considered in such reserves. As discusses in Section 5.1.4, it is very difficult for to incorporate many of the larger-scale ecological processes into the average reserve. A reserve cannot be seen in isolation from the landscape in which it exists. There are no statutory reserves on this planet that would retain their initial biodiversity component in the long term in isolation from their surrounding landscape. To accommodate all processes it is imperative that planning be conducted at the landscape level and consider all forms of land-use and how these contribute to achieving different biodiversity targets. How a region achieves both its biodiversity pattern and process targets in terms of land-use allocation needs to be interpreted within the context of the region and mechanisms available for achieving those targets.

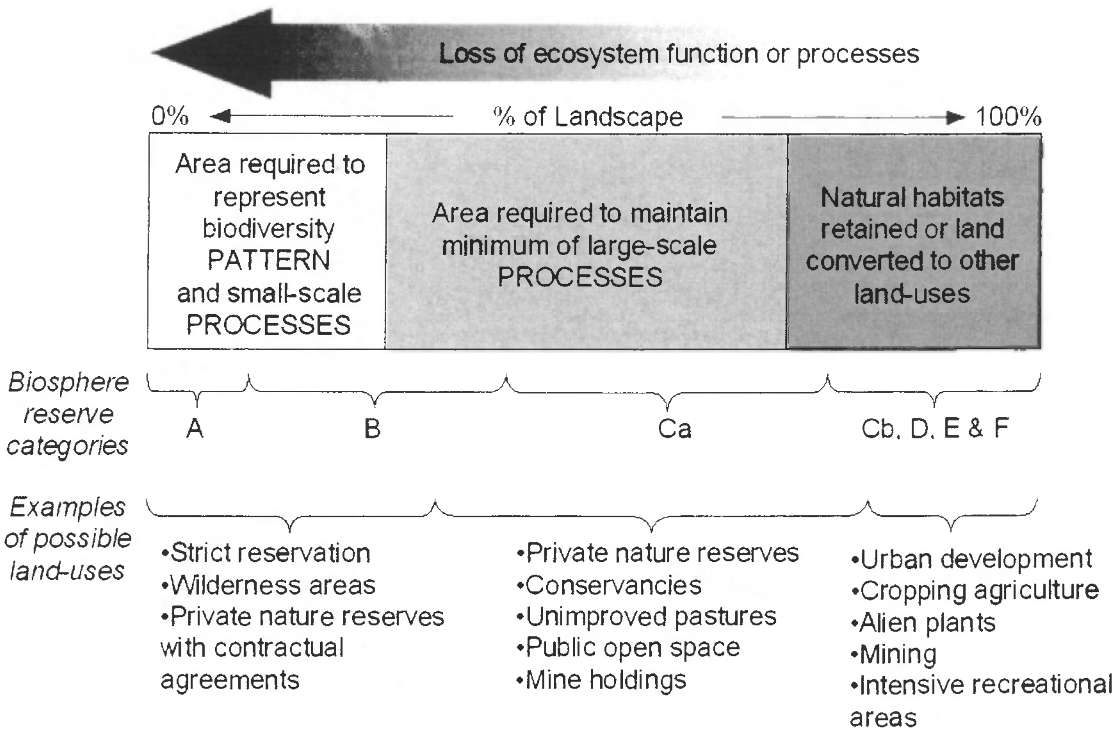


Figure 5.11: The potential relationship between conservation targets, the biosphere reserve and land-use.

5.3.1 Core Reserve Goals

The primary goal of the core Knersvlakte reserve is to conserve a representative sample of the unique flora and habitats of the Knersvlakte region. Of primary interest here is conservation of the quartz-patch, limestone and quartzite mountain habitats and their associated biota. These habitats have been the focus of conservation concern in the region over the years (Hilton-Taylor 1994b). This focuses attention for creating such a reserve in the central area of the planning domain where the majority of these habitats are located.

It is unrealistic to think that a single reserve will satisfy all conservation targets for the planning domain. As is evident in Section 5.2.1, biodiversity is distributed throughout the planning domain. If our goal were to represent all biodiversity features in the planning domain in a reserve of some kind, then this would require a larger extended reserve network with core conservation areas distributed across the entire planning domain. At some point in the future this will need to be addressed in detail, although this goal partly addressed here by the biosphere reserve.

Thus the goal of the core reserve is to maximise achieving pattern targets. In other words, attempt to include as much of this unique biodiversity pattern within a single statutory reserve at the expense of meeting large-scale process targets. As discussed previously, experience from the previous Knersvlakte study and the CAPE study showed that incorporating large-scale processes in the formal reserve network is land-hungry and often requires that one includes large amounts of biodiversity features over and above their pattern targets or non-targeted biodiversity features in order to meet the process targets (Desmet *et al.* 1999; Cowling *et al.* 2003b). The rationale for targeting ecological processes is to ensure that biodiversity persists. It is not, however, a prerequisite that these processes are conserved in statutory conservation areas. Although, many processes are included by default within a medium sized reserve of between 50 000 to 100 000 ha (Appendix 5.5). Also, given the larger scale of some of the processes considered it is impractical to consider incorporating them into statutory reserves. Any biodiversity compatible land-use that allows these processes to persist will meet the process conservation goals.

This shift from balancing the trade-off between achieving pattern versus process targets to a core reserve design skewed towards achieving pattern targets is a major departure from the conceptual approach adopted in the previous Knersvlakte study. The short-term benefit of this "stamp collecting" approach is that it maximises the amount of biodiversity represented within the statutory core reserve. The long-term disadvantage of this approach is that it is imperative that the statutory reserve be nested within an extended "reserve" network or bioregional land-use framework such as the biosphere reserve. Without the surrounding buffer and corridor areas the core reserve will experience a slow loss of biodiversity as large-scale processes are slowly eroded. In this study, large-scale and long-term processes that require much larger areas to persist are specifically targeted at the landscape scale in the design of the biosphere reserve and not explicitly in the core reserve.

5.3.2 Core Reserve Targets

A single reserve cannot conceivably achieve targets for all biodiversity features in the planning domain. Biodiversity is distributed throughout the planning domain, as it does everywhere, and a single statutory reserve that attempts to represent this pattern within a single reserve would have to stretch the length and breadth of the planning domain. Also, the previous Knersvlakte study identified the central area of the planning domain centred on the Sout River as being the centre of diversity of the quartz patch and limestone habitats. This area is roughly the triangle between Vanrhynsdorp, Vredendal and Bitterfontein. This central area is the focus of the core reserve. Thus, for ease of analysis the feature set used to design the core reserve was restricted to include all the focus habitat types, as these are the major focus of the reserve, and only those vegetation types and land-classes that have more than 50% of their global extent within this central area. In other words those features that have a high probability of having their targets met within a reserve that focuses on maximising focus habitat type targets achieved. Using all features in the analysis would be confusing during the design process as the targets for most features would never be addressed leading to brightly coloured irreplaceability maps that would never change. Thus, the core reserve used only 23 out of a total of 124 biodiversity features in the complete dataset to help design the core reserve (Table 5.8). Targets were, however, set for all 124 features (Appendix 5.6).

Targets for the SKEP vegetation types were set to those used for the SKEP Project (Table 5.8, Appendix 5.6). These targets were developed using the species-area method discussed in the Chapter 3.

Unlike the vegetation types, the land-classes have been standardized for biodiversity turnover. Whereas vegetation types can be fairly heterogeneous internally, leading to the variable targets set, this variability has been controlled for in the mapping of the land-classes (Ferrier 2002). Therefore, a single target can apply to all land-classes. The average target for SKEP vegetation types was used for land-classes (Table 5.8, Appendix 5.6). Land-classes are surrogates the same plant biodiversity, but simply mapped in a different manner. Thus, the total area required meeting all vegetation type targets would the same as the total required for land-classes.

Table 5.8: The 23 biodiversity features used to in the design of the core Knersvlakte reserve. Targets are expressed as a percentage of each features original extent in the planning domain and not their global extent. See text for explanation of how each group of features targets were developed.

Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
SKEP Vegetation types				
Northern Knersvlakte Lowland Succulent Karoo	113012.5	112939.8	35	39554.38
Nuwerus Quartzite Succulent Karoo	62456.75	61473.5	35	21859.86
Knersvlakte Quartzfields	122444	121247	40	48977.6
Central Knersvlakte Lowland Succulent Karoo	16768.5	16652.25	35	5868.98
Rooiberg Quartzite Succulent Karoo	16598	16473.75	35	5809.3
Namaqualand Spinescent Grasslands	49487.75	47666.25	35	17320.71
Focus Habitat Types				
W quartz patches	2263.25	2069.75	50	1131.63
W intermediate heuweltjie/quartz veld	1252.5	1218.25	50	626.25
N quartz patches	11383.5	11203.25	50	5691.75
N intermediate heuweltjie/quartz veld	5248.25	5244.75	50	2624.13
Central quartz patches	13130	13105.5	50	6565
Central intermediate heuweltjie/quartz veld	5345.75	5345.75	50	2672.88
Limestone	4920.75	4776	50	2460.38
SE intermediate heuweltjie/quartz veld	757.25	681.5	50	378.63
SE quartz patches	1267.75	1000	50	633.88
SW quartz patches	421.75	190	50	210.88
SW intermediate heuweltjie/quartz veld	32.5	3	50	16.25
Landclasses				
Value-2	87.5	74.75	33	28.88
Value-13	49369	48749.5	33	16291.77
Value-29	45821.5	45276.25	33	15121.1
Value-40	75770.75	70020.5	33	25004.35
Value-43	4923	4666.5	33	1624.59
Value-66	14506	10084	33	4786.98

The targets used for focus habitat types were those used in the original Knersvlakte study (Desmet *et al.* 1999) (Table 5.8, Appendix 5.6). The target for these habitats was set to 50% of their original extent. There is no justifiable rationale for this target other than these habitats are globally unique and the flora that occurs there is found only in the Knersvlakte and nowhere else. Their occurrence in the planning domain represents the only opportunity for conserving these habitats and associated biodiversity. Given their unique status it may be justifiable to consider these habitats on a par with wetlands and forests in South Africa and target 100% of these habitats as any transformation or degradation invariably leads to a permanent loss of these habitats.

5.3.3 Reserve design rules

Irreplaceability provides a quantitative measure of each site's contribution towards achieving the conservation targets discussed in the previous section. In transformed landscapes where the majority of the remaining natural habitat is required to achieve targets most sites will have an irreplaceability of 1 and the options for conservation are limited to where this remaining habitat is located. In relatively untransformed landscapes, such as the Knersvlakte, the options for achieving targets can be numerous and irreplaceability does not explicitly say which sites should be selected to design a reserve. In the design process it helps to incorporate a set of design rules that guide how a reserve is constructed or where it is located in the landscape. These rules can incorporate ecological consideration such as a single large reserve or minimise the reserve edge relative to area. These rules can also incorporate practical or management criteria such as avoiding certain land-use types or elements of human infrastructure such as roads.

In the previous Knersvlakte study design rules included ecological considerations (include at least three adjacent and complete drainage basins) as well as management criteria (limit the reserve extent to less than 50 000ha and avoid including the Saldanha-Sishen railway and N7 national road in the reserve). Subsequent research has shown that the original drainage basin hypothesis, which as a surrogate for evolutionary processes, was not valid and it is rather the fine-scale juxtaposition of quartz-patch types that is driving speciation in the quartz-patch flora (Ellis 1999). Also, the management design rules specified by the SANP fell away as the reserve is no longer earmarked for development as a national park. These management design rules severely constrained the configuration of the reserve as the national road and railway line quarters the core area of diversity. Options for reserve establishment were effectively limited to choosing that quarter of the central area that achieved targets best.

In this study the reserve design rules have changed to reflect the new ecological information and context within which this design process is being conducted. The previous management design rules are no longer considered as none were specified by the implementing agency (WCNCB). The following set of design rules were developed to assist with selecting sites for inclusion in the core reserve:

- Design a single contiguous reserve between 50 000 and 100 000ha in extent;
- Maximise the number of biodiversity pattern features, especially focus habitat types, included in the reserve;
- Include the centre of quartz-patch and limestone habitat diversity associated with the Sout River;
- Where possible include upland-lowland and biogeographical gradients;
- Include the existing reserved property of Moedverlooren.

The reserve size rule is based on an analysis of the size of reserves (SKEP category 1) in SKEP planning domain. As no size specifications were provided by the WCNCB it was assumed that this reserve would be a flagship reserve of a similar size to other large reserves currently managed by the WCNCB (Figure 5.12).

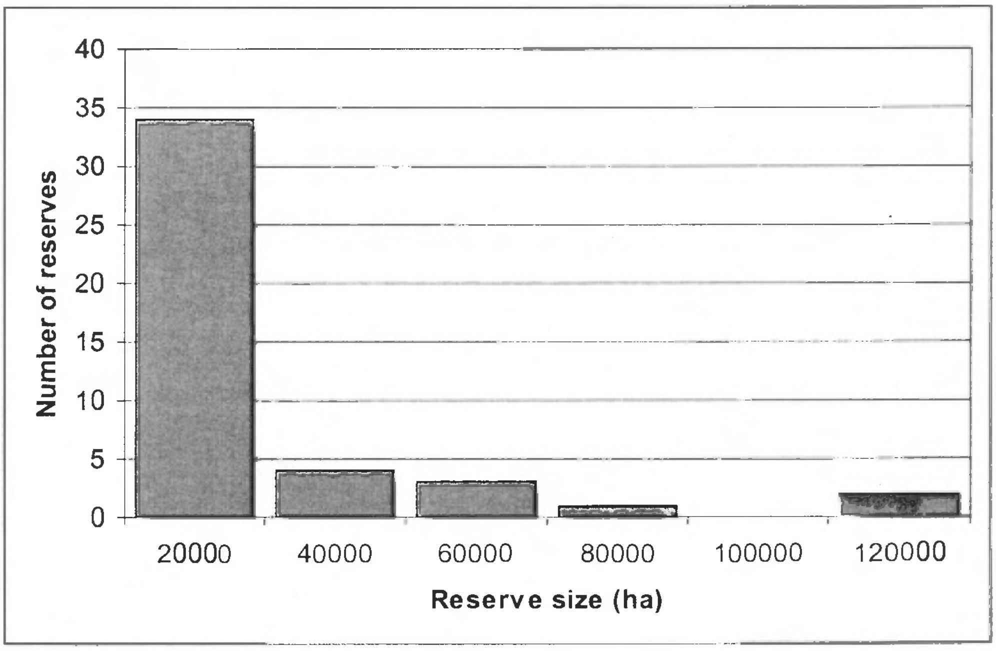


Figure 5.12 The size of SKEP category 1 reserves in the SKEP planning domain. The majority of the SKEP planning domain in South Africa fall under the juristiction of the WCNCB.

5.4 Step 3: Review the efficacy of the existing reserve network (GAP Analysis).

The existing reserve network does a poor job at representing the biodiversity pattern of the planning domain (Figure 5.13 & Figure 5.14; Appendix 5.7). All statutory reserves and proclaimed conservancies are considered here to contribute to the reserve network. Nearly half the features targeted are not represented in any reserve at all (Figure 5.13) and only one feature (land-class 74) is represented within a reserve that meets the target set for that feature (Figure 5.14). A significant amount of work is required to create a representative reserve network in the region.

Figure 5.13: The percentage of the original area of each feature already conserved within the existing reserve network summarised for the 124 features biodiversity pattern (focus habitat types, vegetation types and land-classes) in the planning domain. Only five features have greater than 10% of their area within existing reserves whereas 56 are not represented within any reserve at all. Sixteen land-class features with occurrences of <50ha are not considered here. The numbers opposite each segment represent the total number of features in each category.

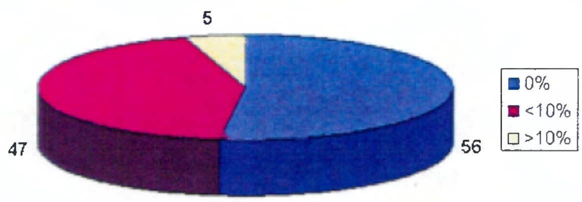
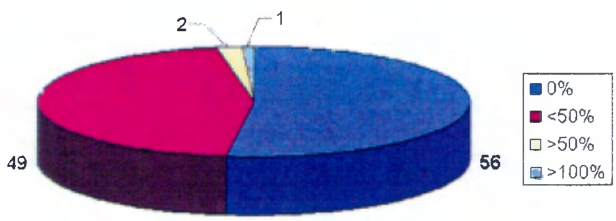


Figure 5.14: The percentage of each feature target achieved within the existing reserve network summarised for the 124 biodiversity pattern features (focus habitat types, vegetation types and land-classes) in the planning domain. For example, only one feature has its target achieved in the existing reserve network and two have greater than 50% of their target achieved. Sixteen land-class features with occurrences of <50ha are not considered here. The numbers opposite each segment represent the total number of features in each category.



5.5 *Step 4: Review Alternative Land-Use Options.*

This step of the planning process involves the identification of types, patterns and rates of alternative land-use processes that could compromise the achievement of the conservation goals set out for the region. In the Knersvlakte, this amounts to assessing the potential for areas to support the three most important alternative land-uses of the region, namely livestock grazing, cropping agriculture and mining (Cowling *et al.* 1999a). The likelihood that a site will be converted to a non-biodiversity compatible land-use, viz. the site's vulnerability, such as cropping agriculture or mining is a key indicator used to schedule conservation action. Areas of conservation importance that have a high vulnerability to being transformed within the time framework assessed are the areas that require immediate conservation action. Areas of conservation importance that have a low vulnerability are not in immediate danger of being transformed; therefore, the scheduling of conservation action need not be an immediate priority. The rationale for performing such an analysis relates to increasing the retention of biodiversity by scheduling conservation action such that the risk of loosing biodiversity is minimised.

In this study only agricultural and mining potential are assessed. For both land-uses an assessment of vulnerability is assessed on the basis of existing resource potential data obtained from various sources. No attempt is made here to develop more predictive models, or incorporate economic or social data to estimate the rate or potential of either land-use to spread. Well-managed livestock grazing is viewed as being compatible with the conservation of biodiversity and is therefore not included in this analysis. Exceptions will always occur.

5.5.1 Agricultural Potential

Cropping agriculture presents the greatest agricultural conflict with biodiversity conservation. The conversion of land to agricultural fields permanently transforms natural habitat. These "holes" in the natural fabric of a landscape are essentially valueless to the majority of biodiversity. Thus, landscape planning needs to trade off biodiversity goals with agricultural development requirements.

Over much of the planning domain, the development of dryland cropping is static. Given the arid climate and the absence of government subsidies dryland cropping will remain limited to existing croplands with further expansion of croplands unlikely. The development of irrigation cropping has great potential in the planning domain especially on the deeper, non-saline Sandveld and aeolian sand derived soils. This form of cropping is, however, limited by the availability of further water resources being made available either from the Doring or Olifants Rivers. Thus, the development of irrigation cropping is limited to the areas adjoining the Olifants River down stream of the Doring River confluence.

In the previous Knersvlakte study (Desmet *et al.* 1999) a 1:250 000 scale agricultural potential map from ARC covering the entire planning domain was used to assess agricultural potential. In this study uses a 1:50 000 scale agricultural soil potential map for the lower Olifants River region that was developed by J. Lambrechts (jjnl@sun.ac.za) and B. Schloms (bhas@sun.ac.za) at the University of Stellenbosch as part of the Wider Olifants-Doring Irrigation Scheme environmental scooping study conducted by Arcus-Gibb consulting engineers. The coverage of this soil potential map is much narrower than the previous agricultural potential map used, however, the spatial accuracy is higher and the area of interest is that most likely to be affected in the future by cropping agriculture.

The soil potential map ranks soils in terms of their potential to support tuberous (i.e. annual) and non-tuberous (i.e. perennial) crops relative maximum possible yields on ideal soils. The values in each field are the percentage of maximum potential crop yield. For tuberous crops values range between 0 and 80%; and, for non-tuberous crops between 0 and 75%. For the analyses, the soil potential values were ranked into four classes – none (0%), low (>0% to 40%), medium (>40% to 60%) and high (>60). The cutoffs between each category were determined with advice from B. Schloms (Pers. Comm.). By combining the ranks for the two crop types a single cropping agricultural potential layer was developed. For each polygon the combined rank was set to the higher of the two crop-type ranks. For example, if the tuberous crop ranked for a polygon were low, and non-tuberous crops high, then the combined rank would be high.

The Wider Olifant-Doring Rivers Irrigation Scheme (WODRIS) soil potential was merged with the original ARC agricultural potential map to create an agricultural potential layer that covers the majority of the planning domain. Any area identified in the ARC map as suitable for irrigation agriculture were ranked as low in the merged map. Any agricultural development in these areas, i.e. outside the environs of the lower Olifants River, is limited by the availability of water.

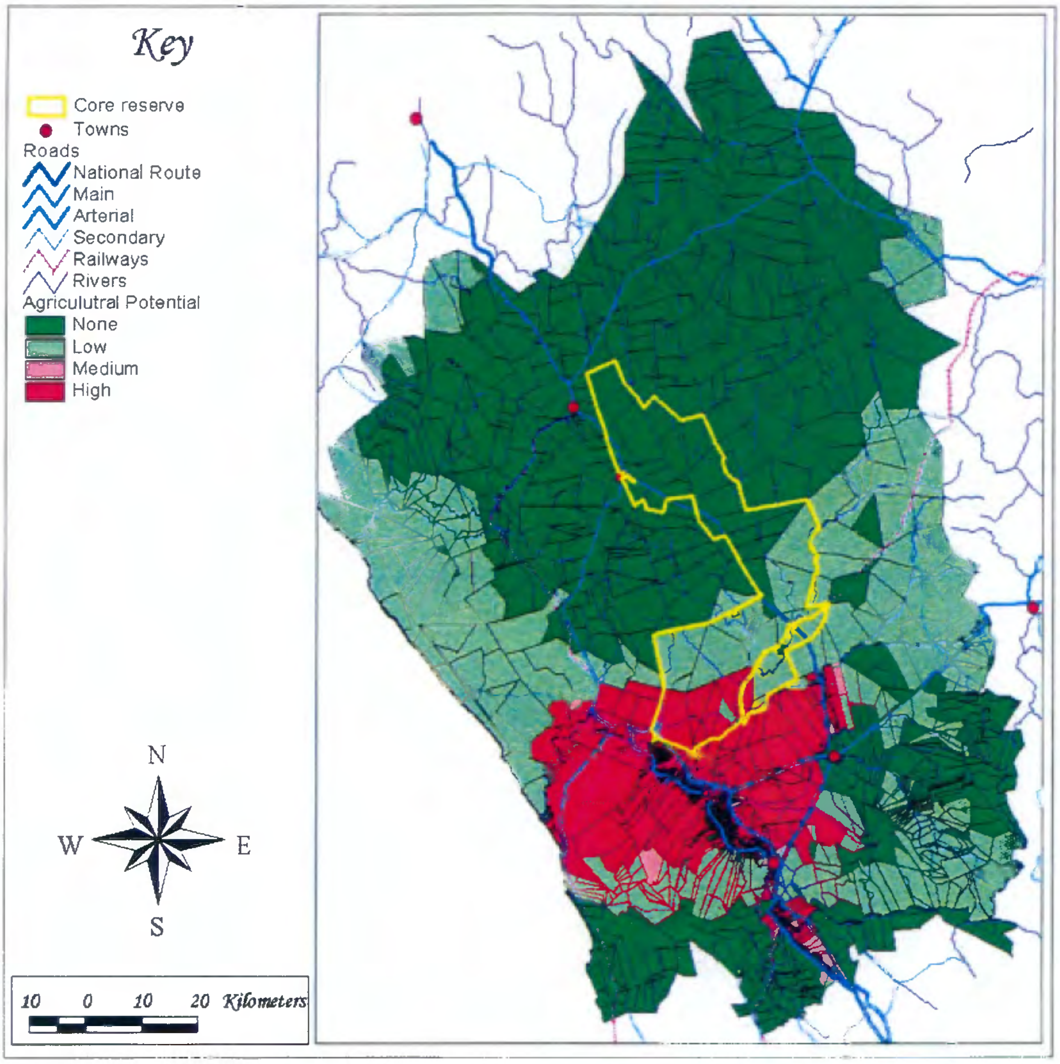


Figure 5.15: Cropping agriculture potential of cadastres in the planning domain. The proposed boundaries of the revised core reserve are indicated in yellow on the map.

The agricultural potential layer was intersected with the cadastral boundary layer for the planning domain and the area of each agricultural rank category summarized by cadastre. Each cadastre was then assigned an overall agricultural potential rank based on the composition of resource fields. A cadastre was assigned the rank of the highest occurring resource field rank if greater than 10% of the area of the cadastre or 10 ha of that agricultural potential rank occurred in the cadastre. If these conditions were not met then the cadastre was assigned to the next highest agricultural potential rank occurring in the cadastre (Figure 5.15).

5.5.2 Mining Potential

Like cropping agriculture, mining presents a potential conflict with biodiversity conservation goals. Opencast mining, where it involves rocky or quartz-patch substrates, results in the permanent loss of natural habitats. Many of the habitats targeted by mining, such as quartz patches for diamonds and limestone outcrops, are also key habitats for biodiversity. Achieving conservation goals will require a tradeoff with mining development aspirations.

Table 5.9: The extent of the nine mineral resource fields in the planning domain. The six key minerals resources are italicises.

Mineral Resource Field	Area (ha)
Clay	95 303
Diamonds	94 964
Granite	5 167
Gypsum	335 708
Heavy minerals	67 220
Iron	201 835
Limestone	101 190
Marble	46 625
Phosphate	28 028
Silica	1 880
Total area of planning domain covered by all minerals	586 619
Total area of planning domain covered by 5 key minerals	235 391

Mining resource field data for the planning domain was obtained from the Council for Geoscience (Belville). These data are in the form of 1:250 000 scale, broad resource field polygons rather than precise locations of known mineral deposits. Data for a total of nine mineral resources are provided (Table 5.9). Six minerals are identified as key resources based on currently expanding mining activities in the region (heavy minerals, coastal diamonds, limestone, marble, silica) or potential conflict with key biodiversity habitats (inland diamonds, limestone, marble, silica). For the analyses, mineral resources were ranked into four classes based on the occurrence and overlap of resource fields in a polygon. Areas not covered by any resource field are classified as low. Polygons covered by one or more resource field other than the six key minerals are classified as medium.

Areas covered by at least one of the five key minerals are classified as high; and, by two or more as very high.

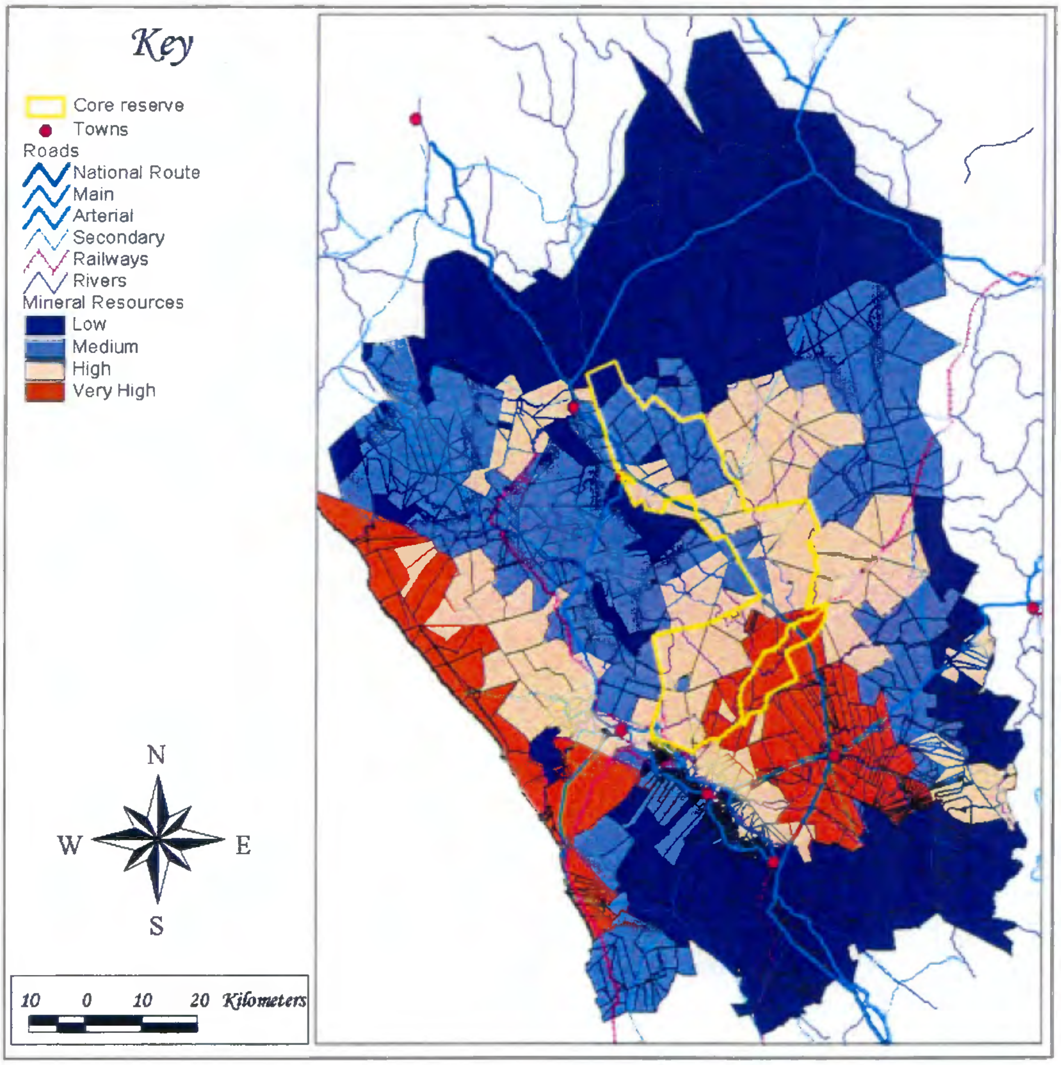


Figure 5.16: Mineral resource potential of cadastres in the planning domain. The proposed boundaries of the revised core reserve are indicated in yellow on the map.

The mineral resource field layer was intersected with the cadastral boundary layer for the planning domain and the area of each mineral rank category summarized by cadastre. Each cadastre was then assigned an overall minerals resource rank based on the composition of resource fields. A cadastre was assigned the rank of the highest occurring resource field rank if greater than 10% of the area of the cadastre or 1 ha of that resource field occurred in the cadastre. If these conditions were not met then the cadastre was assigned to the next highest resource field rank occurring in the cadastre (Figure 5.16).

5.6 Step 5: Designing the Core Reserve.

Step 5 involves the location and design of a core reserve that achieves targets for the selected biodiversity pattern features and the design criteria. Generally, the overall aim of this stage of the planning process is to identify conservation areas that will collectively achieve all the targets for pattern and process. The system of proposed conservation areas might be much larger than the area considered feasible, but sound decisions about the relative importance and urgency of protection for specific parts of the landscape (Step 6) can only be made when the full requirements of all targets have been laid out. As discussed above, this study deals with the design of only a single core reserve in the planning domain and as such the identification of a network of conservation areas that achieves all targets is not addressed here.

In this study, the manner in which the spatial options for achieving the set of conservation targets is mapped, is to calculate and map the irreplaceability of each part of the landscape or planning unit (Pressey *et al.* 1995a). A map of irreplaceability, with values allocated to all planning units, can be considered a map of the options for achieving a set of targets. Areas that are totally irreplaceable are non-negotiable parts of an expanded conservation system, regardless of what form of conservation management is applied (see Step 6). Where irreplaceability is less than one means that there are options for which sites are selected to achieve targets. Where this is the case the design criteria can be applied to make descisions as to which sites to select.

This part of the planning process uses the C-Plan software to map options and assess the contribution of the resultant reserve to achieving the conservation targets. C-Plan does not design the reserve. It is merely a decision support tool for assessing the contribution of alternative reserve scenarios where the number of planning units and biodiversity features considered is large.

Six basic steps in designing the core reserve are followed in this study. These are:

1. Look at the options for all pattern features.
2. Look at the options using a restricted feature-set, i.e. options for the 23 features whose targets are to be achieved within the core reserve.
3. Run a minset to find the most area-efficient way of achieving the targets for the restricted feature-set.
4. Apply the reserve design criteria to locate the core reserve using the minset outcomes as a starting point from which to locate the core reserve.
5. Examine any outstanding targets and find areas to achieve these targets that are in line with the design goals.
6. Repeat steps four and five until the best reserve scenario is achieved, i.e. one that maximizes targets achieved within the bounds of the reserve design criteria.

5.6.1 Core Reserve Design Step 1

Looking at options for achieving targets for all pattern features shows that important areas are distributed throughout the planning domain (Figure 5.17). This shows quite clearly that it will be impossible to achieve all targets for the region in a single reserve. To adequately represent the regions biodiversity within statutory reserves will require a number of reserves spread through the planning domain. Conservation priorities are not just limited to the "core" Knersvlakte area. The relative lack of red sites (i.e. totally irreplaceable, Figure 5.17) means that to a greater or lesser extent there are options for how targets are achieved.

The pattern of higher irreplaceability values in Figure 5.17 already begins to indicate a potential structure for a biosphere reserve in terms of a core area centered in the middle of the planning domain with connections to the coast, the Kamiesberg and the Bokkeveld Mountains.

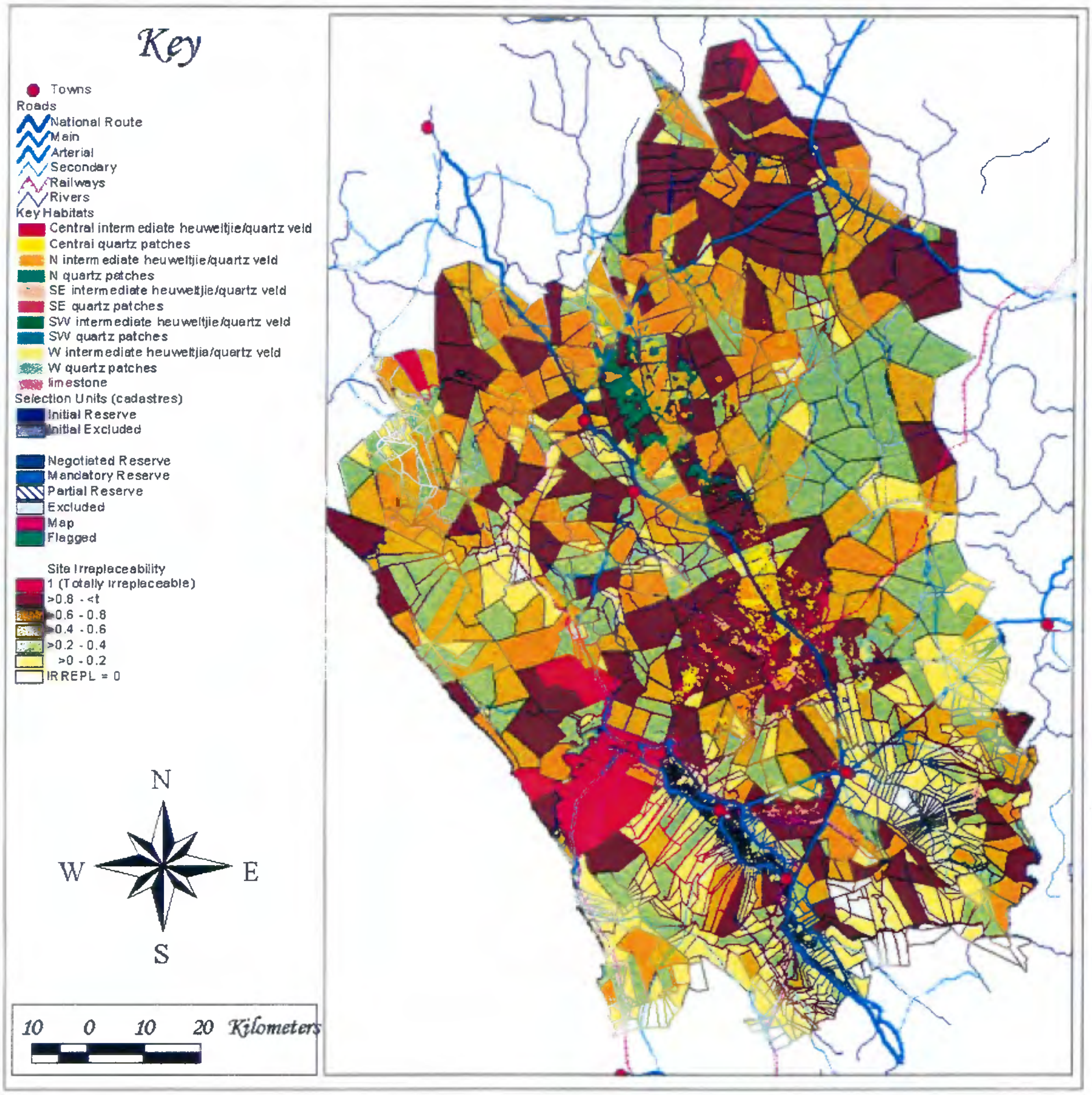


Figure 5.17: Core reserve design step 1 – options for achieving targets using all pattern features.

5.6.2 Core Reserve Design Step 2

By restricting the feature set (see Section 5.3.2) the focus of the reserve design is drawn to the central Knersvlakte area (Figure 5.18). This is expected as the distributions of the features targeted are centered here. In the south, options for achieving targets are more restricted (i.e. more red in the south than north). It is evident that site irreplaceability changes between Figure 5.17 and Figure 5.18 even though the feature extents and targets have not changed. This is because site irreplaceability combines all feature irreplaceability values for a site multiplicatively to produce an index for each site, ranging between zero and one (Anon. 2001). Thus, site irreplaceability will vary depending on the number of features considered. Irreplaceability should not be considered an absolute measure of a sites conservation importance. It should only be regarded as a context-specific relative-measure to help guide conservation or land-use decision.

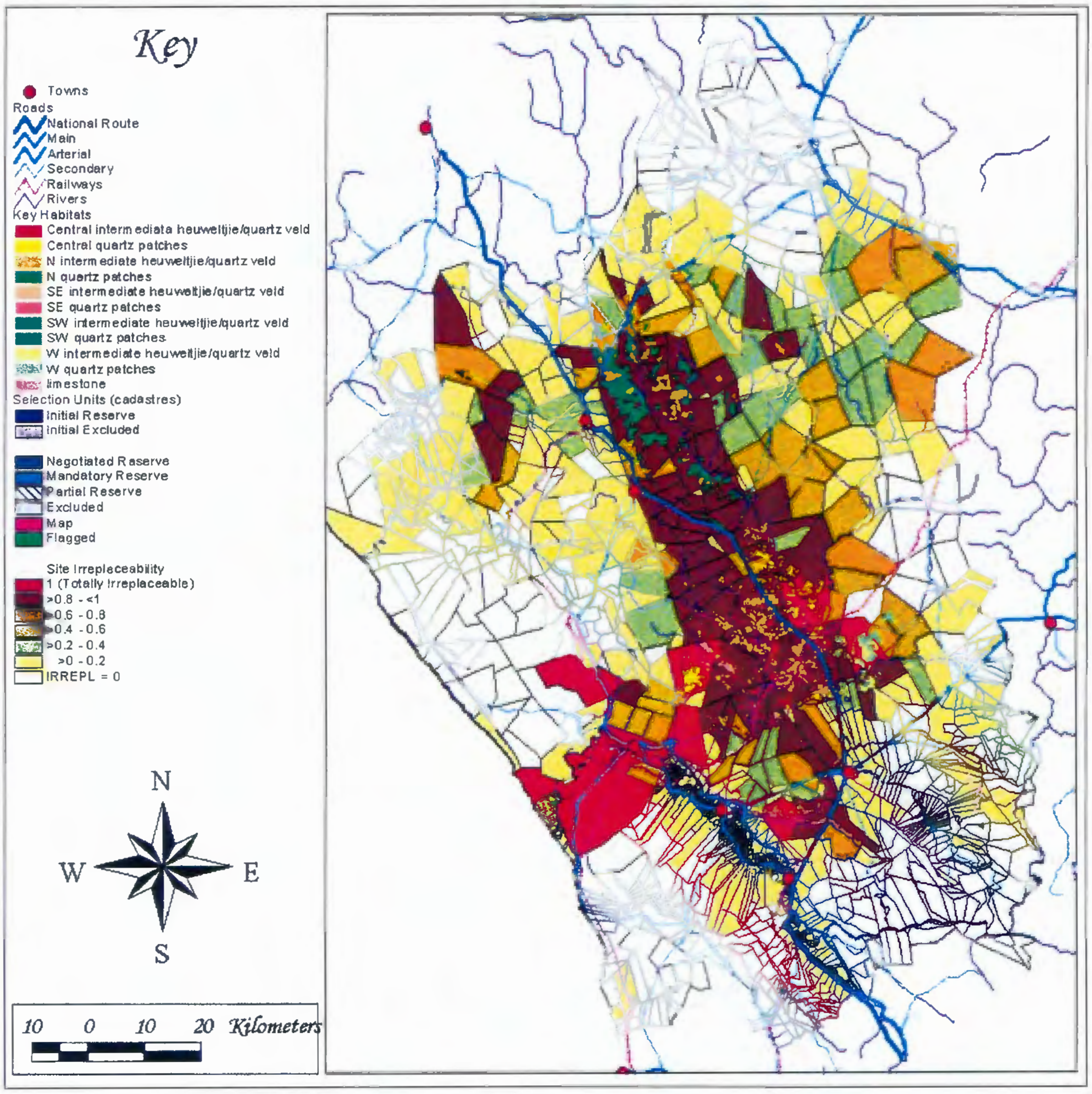


Figure 5.18: Core reserve design step 2 – options for achieving targets using a restricted feature-set.

Figure 5.18 also demonstrates the utility of using a restricted feature-set to design the core reserve. This action helps focus attention on the part of the planning domain where the reserve should be located rather than being distracted by a cacophony of color across the planning domain.

An alternative way to compare site irreplaceability is to ask C-Plan to display the summed irreplaceability for each site. In contrast to the site irreplaceability measure, summed irreplaceability adds the individual feature irreplaceability values (Anon. 2001). Thus, values can range from zero to ten or more depending on the number of features considered. Sites with high values are important for many features whereas sites with values much less than one are not important for any features. The advantage of using summed irreplaceability in a biodiversity rich area such as the Knersvlakte where many sites would be expected to be important, is that is provides a means of discriminating between sites that all have the same or similar irreplaceability values, i.e. two sites can have the same irreplaceability but very different summed irreplaceability (Anon. 2001). Figure 5.19 uses the same feature-set as Figure 5.18 except that site summed irreplaceability is displayed and not irreplaceability. Thus, options for achieving targets become clearer as the large number of brown sites (Irr = 0.8 to <1) in Figure 5.18 are discriminated more clearly in Figure 5.19 to better reflect the individual site contribution to targets.

Viewing options of which sites to select for inclusion in a core reserve is aided by asking C-Plan to display the percent contribution of each site. Percent contribution is the percentage of the total area of each site that would contribute to remaining targets if the site were reserved (Anon. 2001). It is calculated only for mutually exclusive features, i.e. features that do not overlap spatially, in this case SKEP vegetation types. Land-types and focus habitat types are also mutually exclusive features, however, C-Plan can only deal with one set of mutually exclusive features at any one time. Thus, Figure 5.20 shows the percent contribution of sites to achieving SKEP vegetation type targets only. Percent contribution is useful as it is possible to observe how much area of each site contributes to achieving targets. For example, the large site on the southwestern corner of highlighted sites at the mouth of the Olifants River, the Ebenhaeser Colony, has an irreplaceability of 1 in Figure 5.18. However, only a small area of this site contributes to achieving targets.

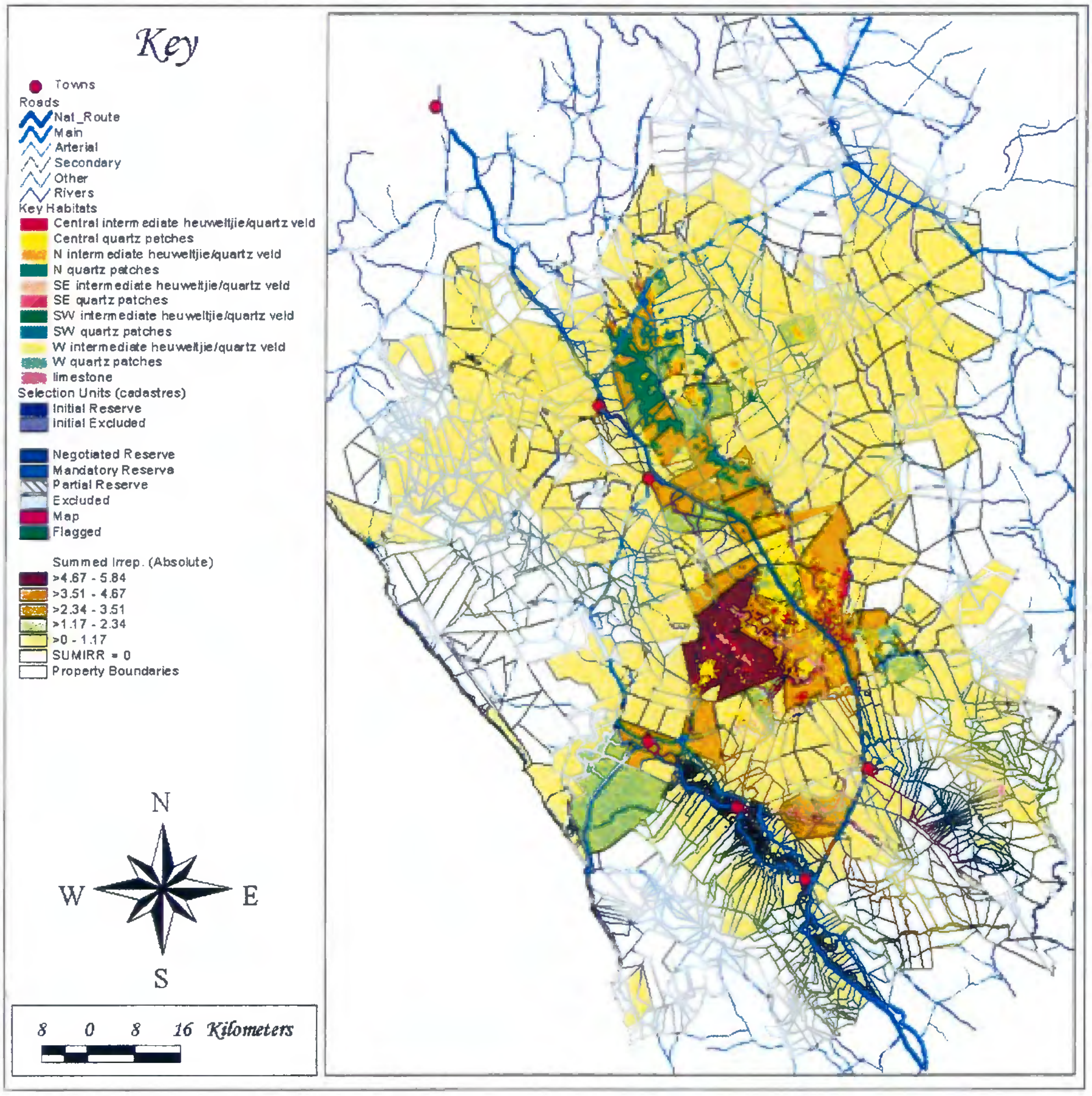


Figure 5.19: Summed irreplaceability for the restricted feature set

After reviewing the initial options for achieving targets, it is now possible to begin selecting sites for inclusion in a reserve. Before embarking on this, a minimum-set is run to estimate what would be the most area efficient means of achieving these targets. In other words what is the smallest area required to achieve the targets. This is a useful exercise as it provides a benchmark against which to measure the contribution and efficiency of the final reserve relative to an "optimal" scenario that minimizes the area required to achieve targets, but which is free from any design criteria.

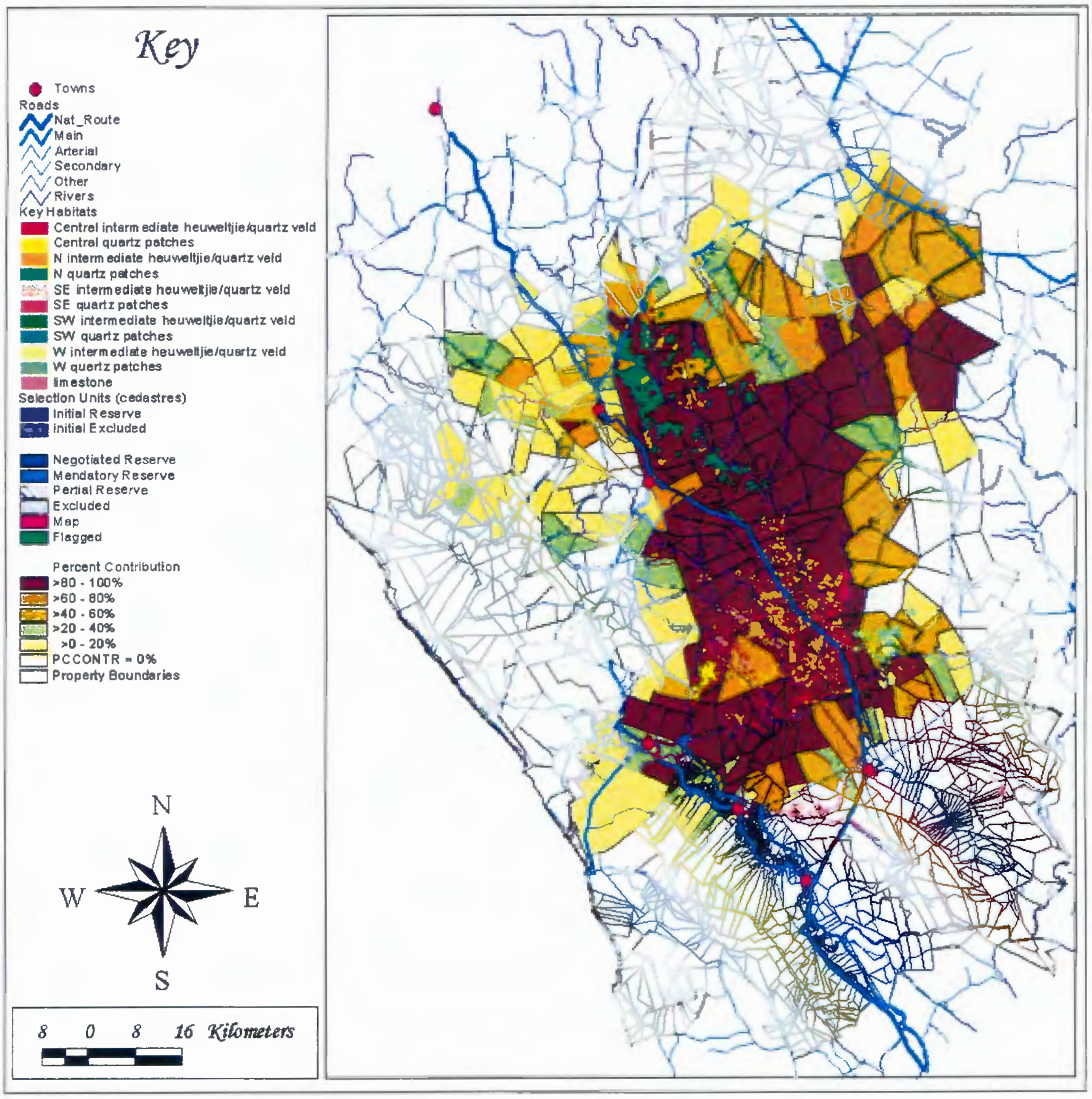


Figure 5.20: Percentage contribution of sites to achieving targets for the SKEP vegetation types only.

5.6.3 Core Reserve Design Step 3

A minimum-set tasked to achieve all targets identified 32 sites (Table 5.10). This could be considered the most spatially efficient solution for achieving the conservation targets for the 23 features in the restricted dataset. Practically, this outcome is not ideal from a reserve management perspective as the reserve is a patchwork scattered across the landscape and it does not meet the design criteria. This design may be useful for the identification of areas in which to locate conservancies across the planning domain.

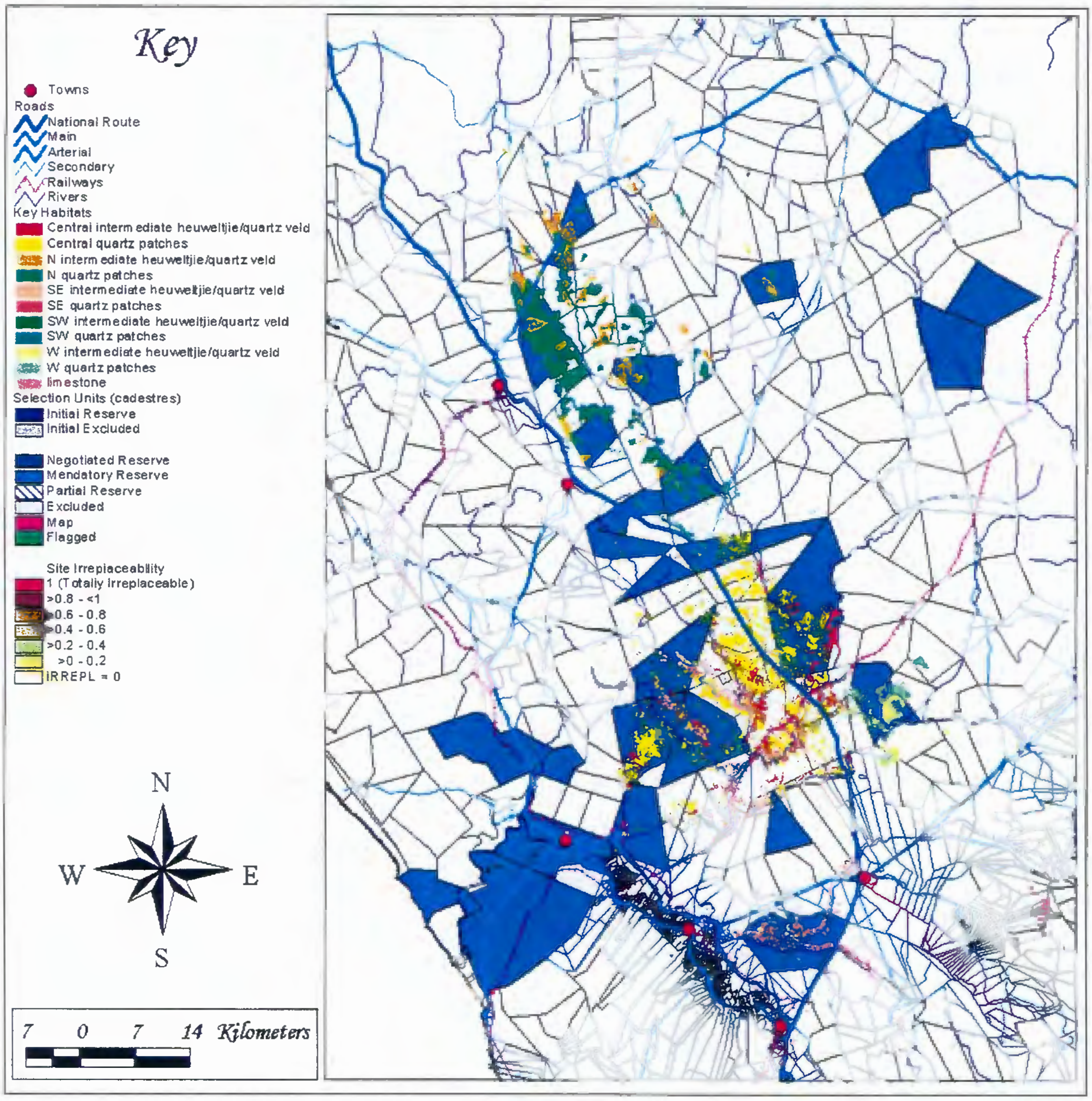


Figure 5.21: Core reserve design step 3 - the outcome of a minimum-set that achieves all targets for the restricted feature-set.

The reserve design process from here to the final reserve outcome is where the reserve design criteria are applied. This is an iterative process and involves removing sites selected by the minimum-set that are located away from the central area of selected sites, and replacing these, if possible, with sites in the center that together satisfy the design criteria as well as contribute to targets. There is naturally a tradeoff here between spatial efficiency and the design criteria. Another tradeoff is that some feature targets may not be met. This happens when a feature does not occur within the central area identified by the minimum set. Some of the southern and western focus habitat types are to remote to be included in a single large reserve.

Table 5.10: Percent of targets for pattern features met by the 32 sites selected in the minimum-set.

Feature	% Of Target Achieved
SW intermediate heuweltjie/quartz veld	100.00
Northern Knersvlakte Lowland Succulent Karoo	100.13
Namaqualand Spinescent Grasslands	100.20
Nuwerus Quartzite Succulent Karoo	103.01
SW quartz patches	103.31
SE quartz patches	104.12
N quartz patches	105.75
Limestone	107.14
Central intermediate heuweltjie/quartz veld	107.86
Knersvlakte Quartz fields	108.63
N intermediate heuweltjie/quartz veld	111.81
Central Knersvlakte Lowland Succulent Karoo	113.02
Central quartz patches	114.23
Value-2	120.00
Value-40	123.17
W intermediate heuweltjie/quartz veld	129.17
W quartz patches	131.38
Value-29	133.95
Value-43	135.77
Value-66	142.69
Value-13	160.86
SE intermediate heuweltjie/quartz veld	184.78
Rooiberg Quartzite Succulent Karoo	276.64

5.6.4 Core Reserve Design Steps 4 to 6

The final reserve outcome illustrated in Figure 5.22 is the result of the iterative process of trading-off design criteria against meeting targets for the restricted feature set. An inevitable outcome of this process is the inclusion of "redundant" sites within the reserve that contribute area for features over and above their targets. These sites are indicated as "map" in Figure 5.22 and are included to maintain the connectivity of the reserve and as such make an important contribution to maintaining ecological processes. It is important to note that although in the target-driven design context these sites are "redundant", from a biodiversity perspective they are still valuable and contain significant amounts of targeted biodiversity features (see summed irreplaceability and percent contribution in

Figure 5.19 and Figure 5.20 respectively). They simply add more area of these features than what was targeted.

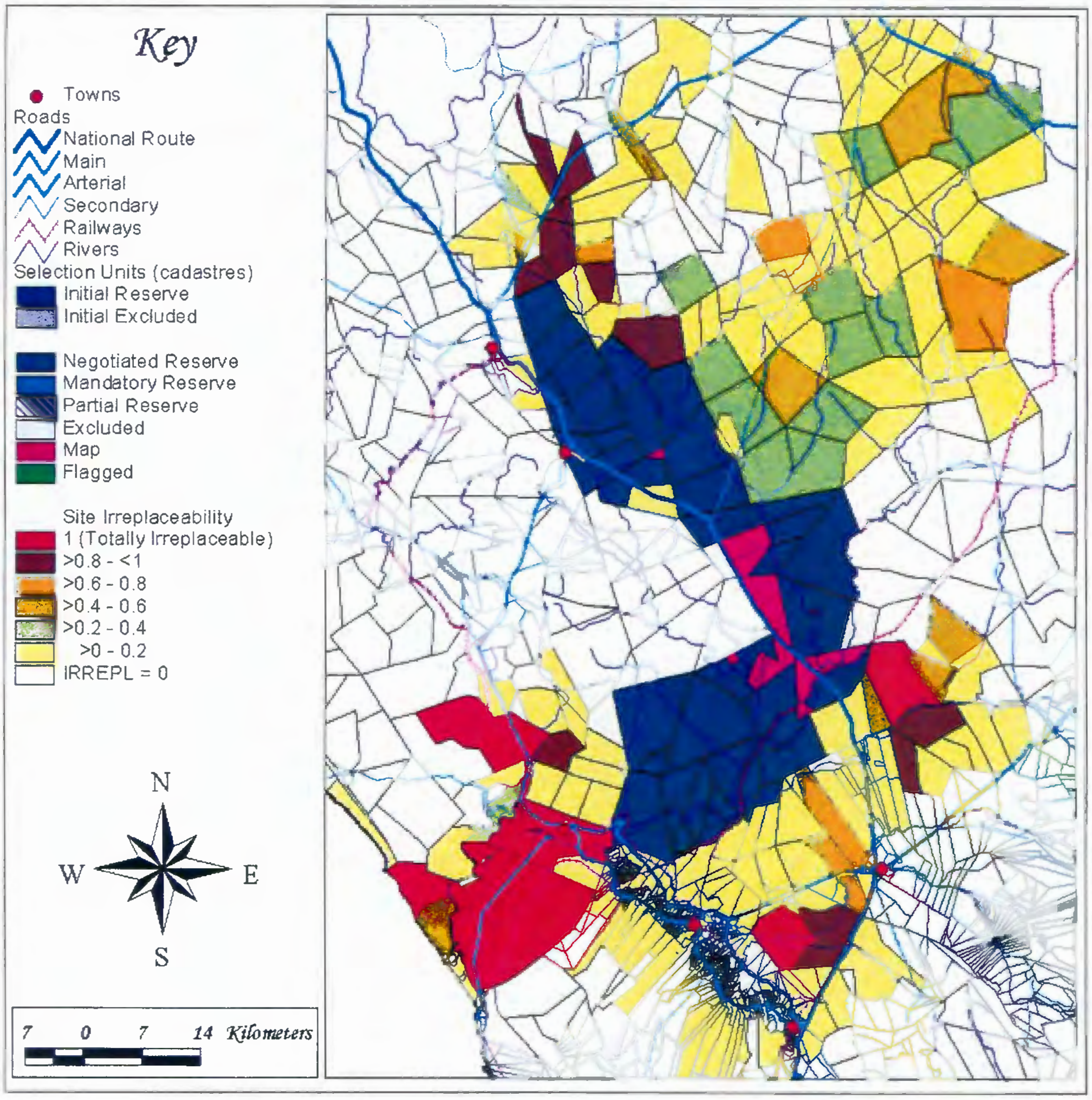


Figure 5.22: Core reserve design steps 4 to 6 – the outcome of applying the design criteria to locate the core reserve. Coloured sites indicate that not all targets have been met with this design.

The colored sites in Figure 5.22 indicate another outcome of this design process. Not all targets can be achieved in this reserve (Table 5.11). For some of the focus habitats it may not be possible to do so in a single reserve. The red sites in Figure 5.22 indicate the mandatory areas required to represent these features in a reserve. Some of these areas can be included in the core reserve by expanding the southern part of the reserve east and west, or the northern part northwards. More remote sites, however, can only be included in separate reserves. All these areas can be conserved with in the context of the biosphere reserve as either buffer or corridor areas, or additional core biosphere reserve areas. This is an important point to note. Should circumstances present themselves that these properties are made available to the reserve then incorporating them into the core reserve will contribute to achieving conservation targets. They are not included in the current proposed reserve design as the present configuration of the reserve satisfies the reserve size design criteria.

Table 5.11: A summary of the percent of initial available target satisfied for the 23 pattern features used to design the core reserve. Features with less than 40% of their target achieved in the reserve are highlighted. These are features that occur away from the core area of the reserve and would require either eastward/westward expansion of the southern part of the reserve or a completely separate reserve if their target is to be met.

Name	Percent of Initial Achievable Target Satisfied
SE intermediate heuweltjie/quartz veld	0.00
SE quartz patches	0.00
SW intermediate heuweltjie/quartz veld	0.00
SW quartz patches	0.17
W intermediate heuweltjie/quartz veld	0.91
W quartz patches	5.05
Northern Knersvlakte Lowland Succulent Karoo	41.09
N intermediate heuweltjie/quartz veld	43.18
Namaqualand Spinescent Grasslands	88.98
Limestone	107.95
Central Knersvlakte Lowland Succulent Karoo	124.17
N quartz patches	152.16
Nuwerus Quartzite Succulent Karoo	159.93
Value-29	160.44
Value-13	187.60
Value-66	195.76
Knersvlakte Quartzfields	200.05
Central quartz patches	220.66
Rooiberg Quartzite Succulent Karoo	223.30
Central intermediate heuweltjie/quartz veld	233.48
Value-2	240.00
Value-40	260.24
Value-43	272.37

Despite the representation focus of the reserve, the large size and layout of the reserve does achieve some medium-scale ecological processes considered in the design criteria

(Figure 5.23). The reserve contains some of the few upland-lowland gradients present in the central Knersvlakte. The three major biogeographic centers of quartz-patch species diversity in the Knersvlakte are captured within the reserve; and, the biogeographical or evolutionary gradients connecting these three centers are captured within the reserve.

In terms of spatial efficiency, the reserve includes 57 cadastres covering an area of 113 473 ha compared to the 32 sites selected by the minimum-set that cover an area of 145 275 ha. The conservation cost of this compromise is that the reserve achieves targets for 14 out of 23 features with an additional three having greater than 40% of their target met in the reserve.

In the southern part of the reserve there is less flexibility around sites selected for meeting targets. Many of the sites included in the southern part of the reserve are mandatory if targets are to be met (i.e. high irreplaceability in Figure 5.18 and Figure 5.19). In addition, the southern part of the reserve has much potential for eastward and westward expansion that achieves targets for focus habitat types (Figure 5.22). There is also potential for similarly important northward expansion of the reserve.

5.7 *Step 6: Setting Priorities for Conservation Action in the Core Reserve*

This step in the planning process involves the scheduling of action for reserve implementation. Scheduling requires that the recommended timing of conservation action should minimise the extent to which conservation targets are compromised before conservation management is applied (Pressey 1997; Lombard *et al.* 1999a). This requires information on the likelihood of forgoing conservation options or loosing biodiversity through alternative land-uses, i.e. site vulnerability from Step 3, and the consequences of this loss or degradation, i.e. site irreplaceability from Step 5. One approach to scheduling of conservation action involves comparing site irreplaceability and vulnerability in two-dimensional space and those sites with the highest irreplaceability and vulnerability are generally considered as the most important conservation priorities (Cowling *et al.* 1999a; Desmet *et al.* 1999; Pressey 1999; Pressey and Taffs 2001a). An alternative approach to setting priorities is to trade-off benefit or irreplaceability against the cost of implementation using a continuous cost surface (Faith and Walker 1996b; Faith and Walker 2002).

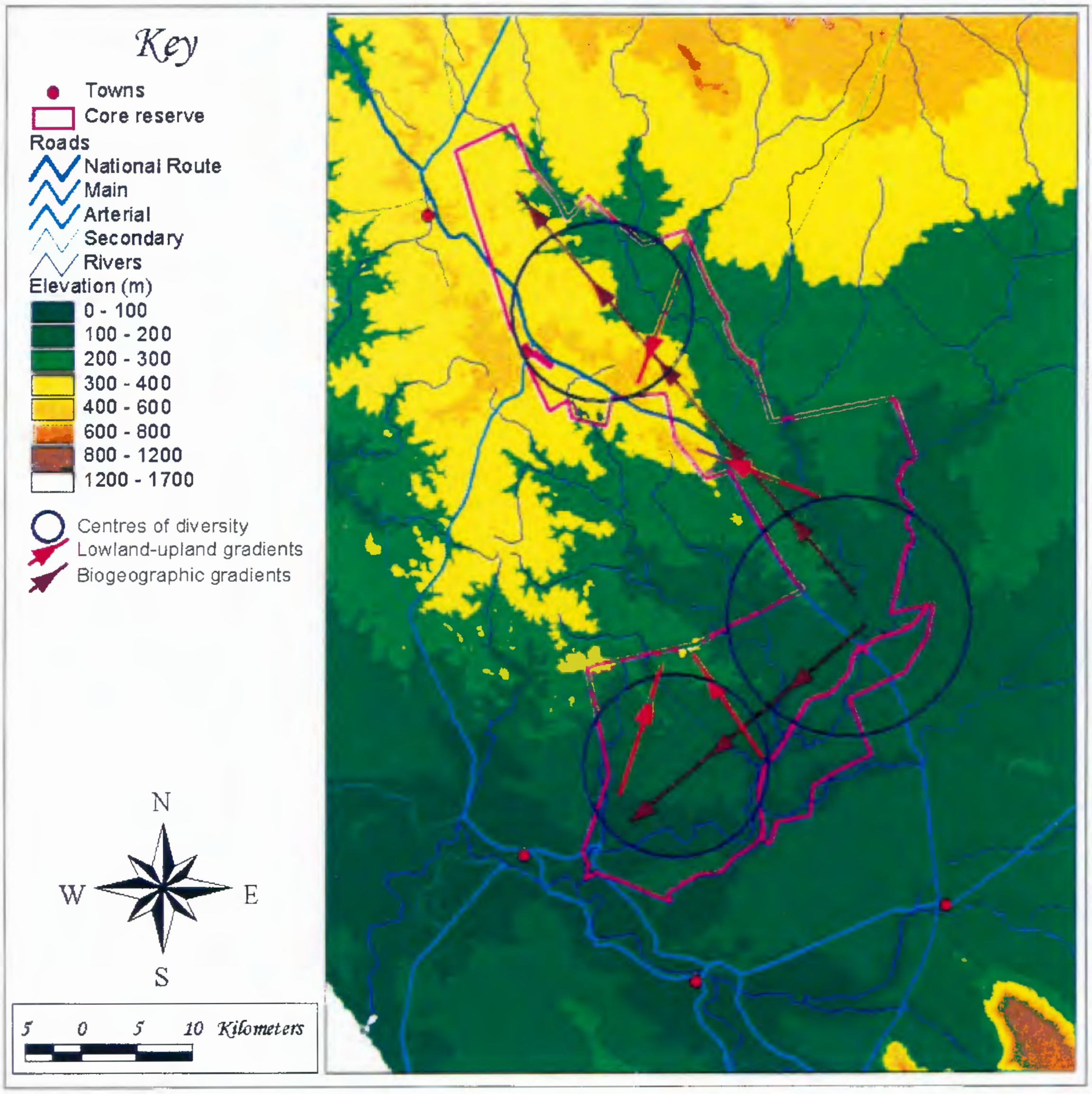


Figure 5.23: Medium-scale ecological processes and biogeographic gradients captured in the reserve.

For the Knersvlakte core reserve the approach to setting priorities for action can be simplified to consider only vulnerability. If the implementing agency agrees with the layout of the reserve design then it follows that all sites identified are equally important from an irreplaceability perspective for ensuring the integrity of the reserve. Once the reserve design has been agreed upon there are no options for how this design is achieved

– all sites are required to create the reserve. All selected sites effectively have an "irreplaceability" of 1. If for example, a property in the middle of the proposed reserve were to be mined for limestone then this would compromise the ecological integrity of the envisioned reserve. Even sites with low irreplaceability based on their contribution to achieving biodiversity targets, but which are required to maintain reserve connectivity, become important in this context. Thus, the scheduling of sites for inclusion in the reserve becomes solely a function of their potential to be lost to alternative land-uses, in this case the potential to be converted to cropping agriculture or mining.

The southern area of the reserve experiences the greatest pressure from competing land-use options whereas the northern area of the reserve is relatively free from these pressures (Figure 5.24). Sites with the greatest potential to support alternative land-uses should be the priority sites for reserve implementation (Figure 5.25). In Figure 5.25 the overall implementation priority rank is based on the combined mineral and agricultural potentials. It is clear that the initial phase of reserve implementation will have to concentrate on consolidating the southern end of the reserve before devoting attention to the north if the reserve is to be created before these alternative land-use options are exercised. It is unlikely that sufficient funds and capacity will be available immediately for the purchase of all properties identified. A more likely scenario is that this process will take several years. Given the patterns of vulnerability and the limited funds available to land acquisition it would be advisable to develop an implementation strategy that focuses on consolidating the southern portion of the reserve before beginning with the northward expansion of the reserve (Figure 5.27). Naturally, should opportunities present themselves in the north before the southern area has been consolidated then it will be up to the LHSKT and WCNCB to decide whether to capitalizes on these opportunities.

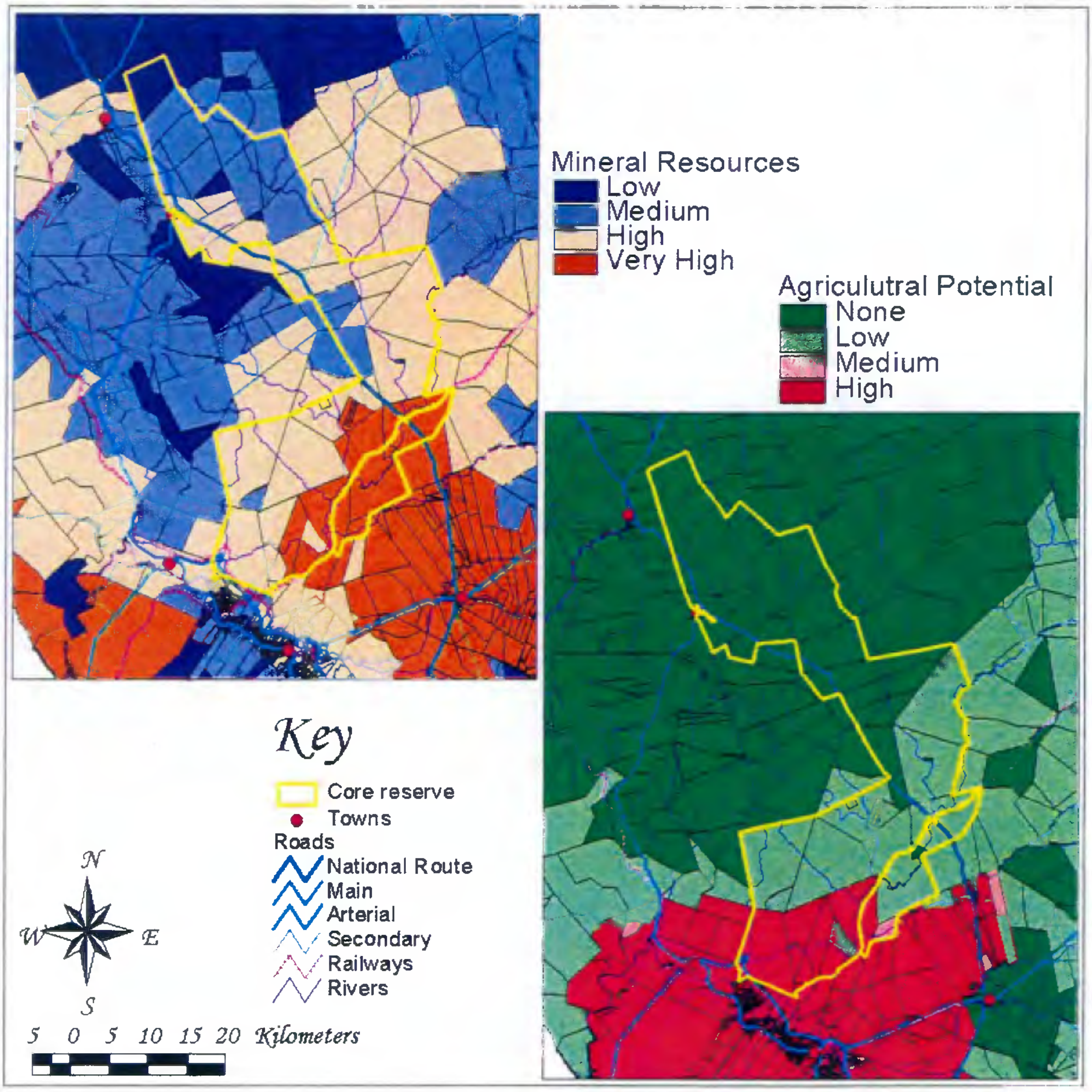


Figure 5.24: Mineral resource and cropping agriculture potential for sites in the core reserve.

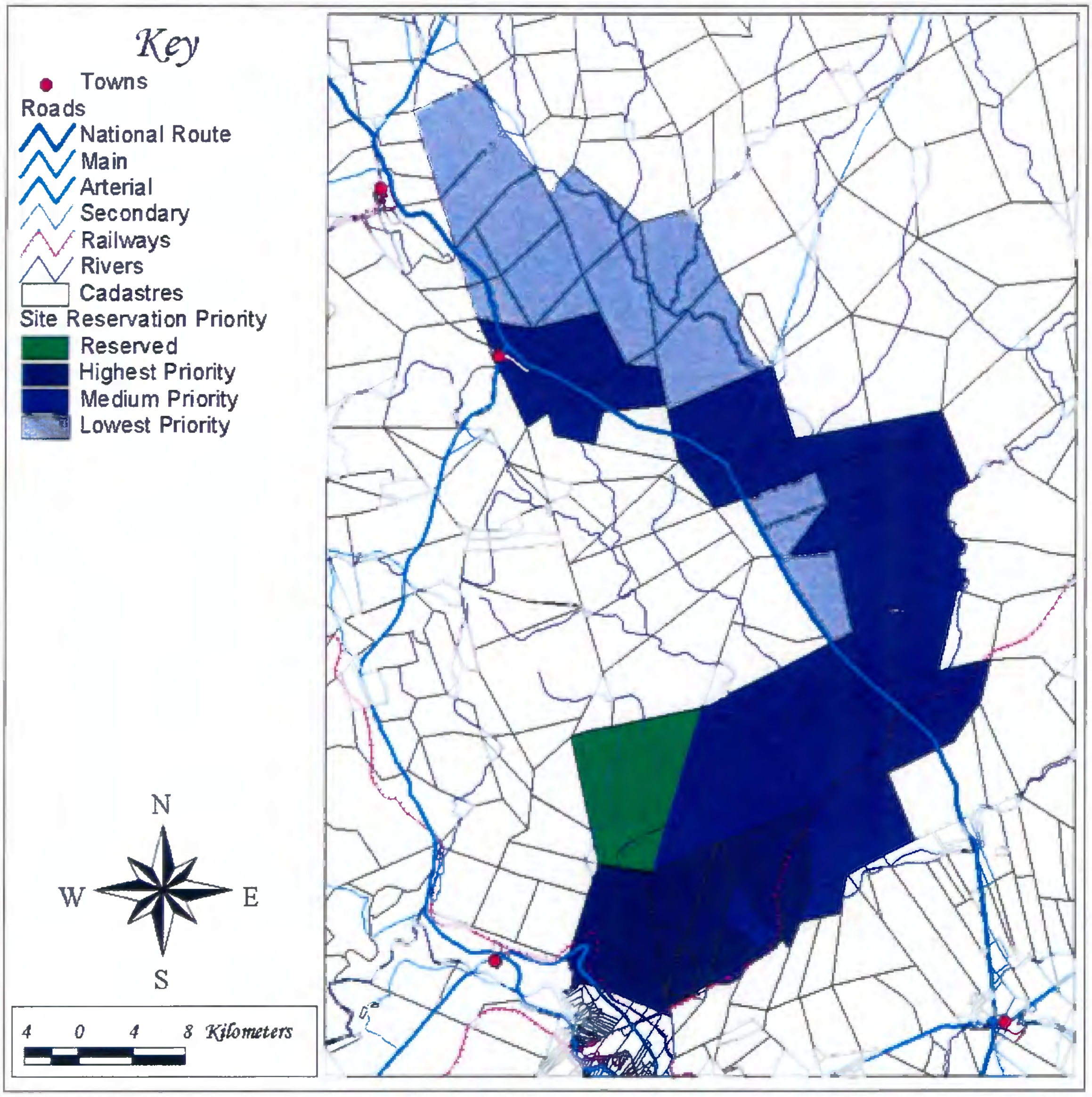


Figure 5.25: An overall priority ranking of sites for inclusion in the reserve based on a combination of each site's mineral resource and cropping agriculture potential.

Based on this vision for an implementation strategy 26 properties are identified for the first phase of reserve development (Figure 5.27). Amongst these properties landownership is divided between eight private individuals or companies; four mining companies; and, the State (Appendix 5.2 and Appendix 5.3). These three categories of ownership will require different strategies if the properties identified are to be included in the proposed reserve. These strategies involve working with the State, mining companies and private landowners.

The State land needs to be purchased as a matter of priority. It is highly likely that the State will sell their land at a fair, market related price, thus helping to correct the distorted land price precedent set with the sale of the farm Moedverlooren. There are land claims on some of these properties. However, no information as to the validity or status of these claims in forthcoming. Should there be successful land-claims on any of these properties, as the agricultural potential of these properties is low the recipients would probably agree to a land exchange for more profitable farming land in the lower Olifants River region. The leader of the Griqua people who are potentially one of the claimants, Dr. Cecile Le Fleur, has already stated their willingness to agree to such an arrangement.

Personal experience of the farmers in the area indicates that all property owners would be willing to sell their properties, but at a price. The sale of Moedverlooren heightened expectations as to what conservation was willing to pay for very marginal agricultural land. This value is a half to double average prices paid for other similar properties in recent years (Figure 5.28). Perhaps a strategy for correcting this perception would be to buy land through the process of "reverse auction". This process involves owners determining the price for their land by competing for the same pool of limited funds for land acquisition made available on an annual basis. The trust would then be able to purchase the best value for money properties each year. This process would require that all landowners be approached and involved in the negotiation process.

A dialogue needs to be initiated with the mining companies concerned drawing their attention to the biological importance of their land. These properties are all owned by large mining concerns and contain significant limestone deposits. It is unlikely that the LHSKT will have sufficient funds to purchase these properties. Recently, a property with a limestone deposit was sold for R77 000/ha (Figure 5.28). Inclusion of these properties in the reserve will require either some form of donation by or contractual agreement with the companies concerned. The SKEP Project has already a pilot mining and biodiversity project involving the Anglo American Corporation in Bushmanland (K. Maze pers. comm.). This project could provide useful insights for resolving this issue. Elsewhere in the Knersvlakte large surface deposits of limestone have almost been eradicated by mining and the deposits remaining in the reserve represent the last opportunity to conserve a large sample of this habitat.

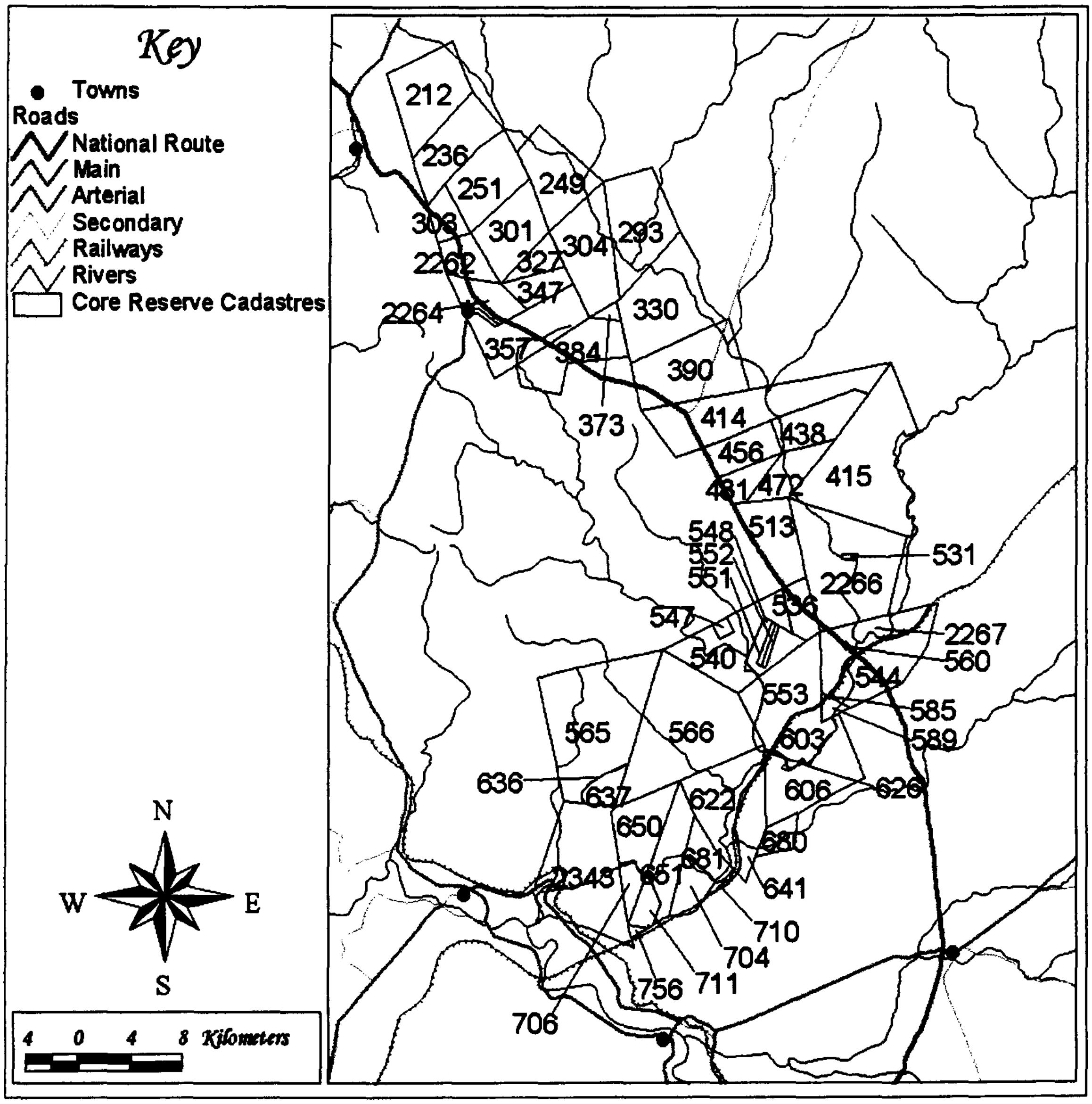


Figure 5.26: The 58 properties identified for inclusion in the Knersvlakte Nature Reserve. Property labels correspond to the Sitekey column in Appendix 5.1.

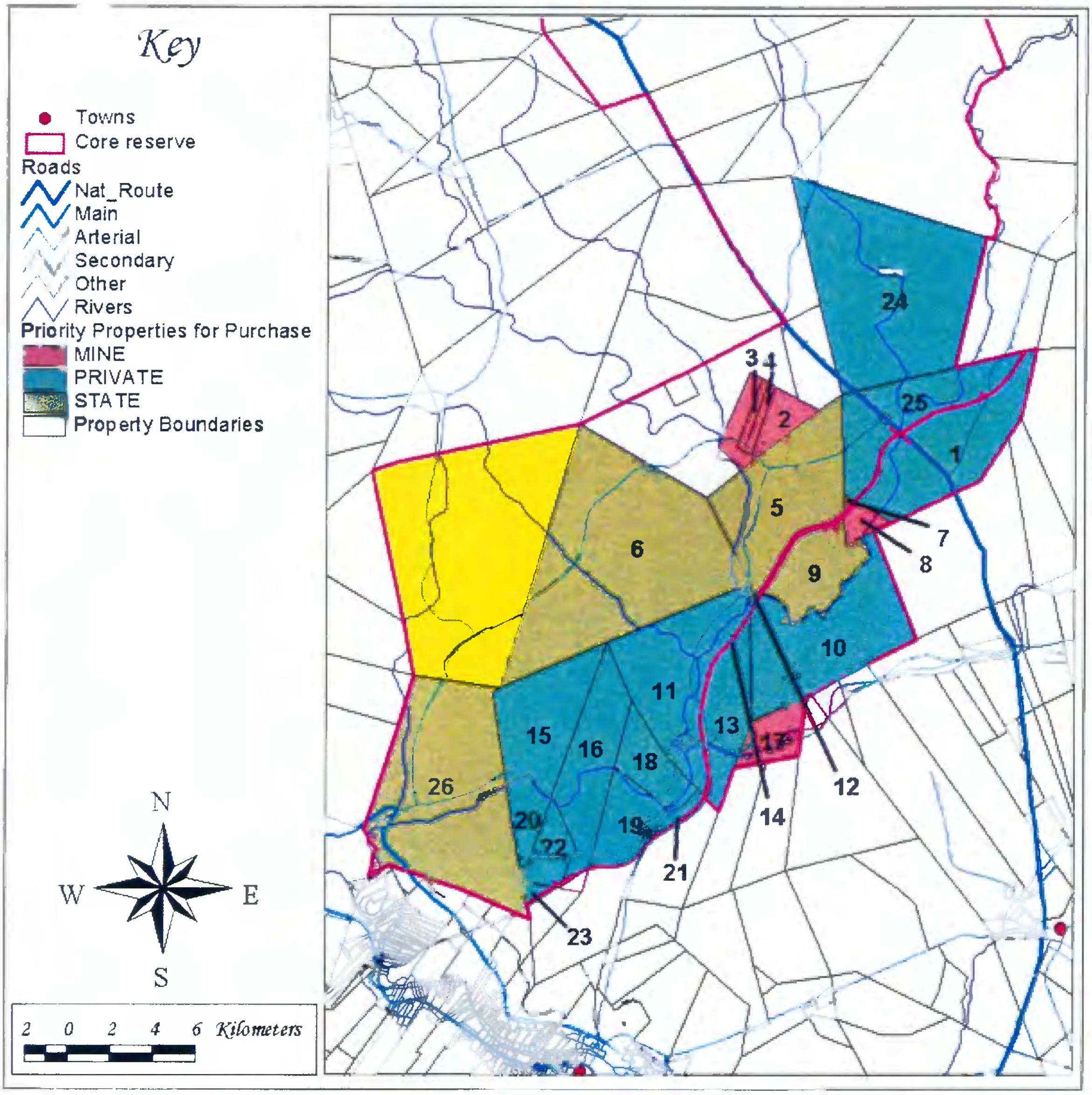


Figure 5.27: Priorities for reserve creation. The 26 properties identified are those that are ranked highest either for agriculture or minerals. The deeds information for each of the numbered properties in this Figure is presented in Appendix 5.2 and Appendix 5.3. The existing "reserved" property, Moedverlooren, is highlighted in yellow. Acquisition of these properties will consolidate the southern boundary of the park as well as form a significant nucleus around which the park can begin to be developed. The northern section of the proposed reserve can be considered for future phases of reserve expansion

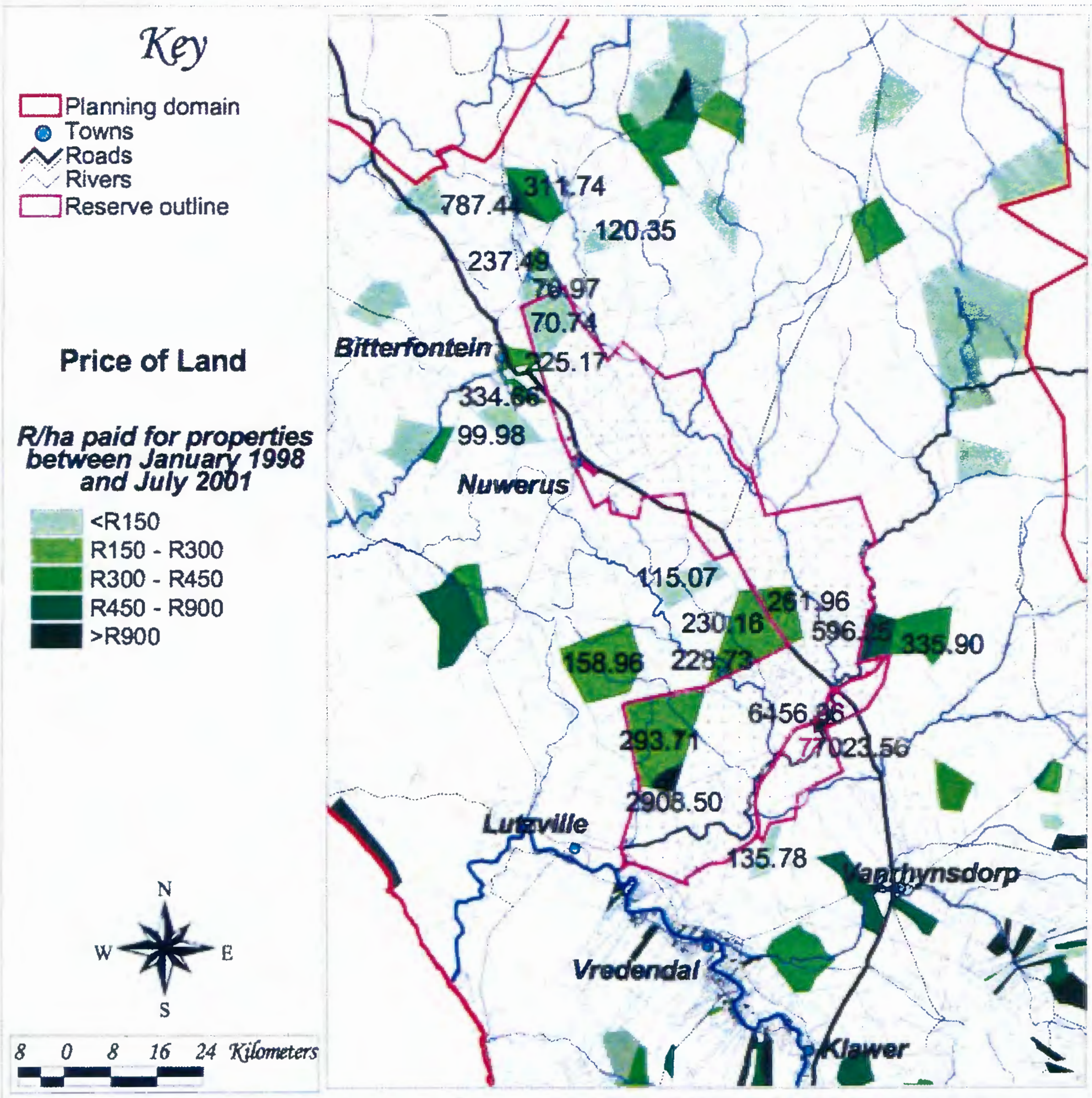


Figure 5.28: Prices paid for properties (R/ha) in the planning domain between January 1998 and July 2001. Text on the figure is the R/ha paid for properties in the immediate proximity to the porposed reserve.

Examination of property prices in the planning domain indicates that market related prices for land should range between R150 and R250 for the southern end of the park and R100 and R200 for the northern end (Figure 5.28). With these values in mind, the present cost of land purchase to establish the entire reserve should range between 17 (R150/ha) and 28 million (R250/ha) Rand. Per hectare prices are variable and, where agricultural potential is constant, this probably relates to farm infrastructure sold with the property or the circumstances of the sale negotiations. Extreme prices in Figure 5.28 (i.e. >R5000/ha) relate to properties sold with economically viable mineral deposits or irrigation cropping potential along the lower Olifants River. Since the compilation of the deeds information in 2001, there have been several properties sold within the area of the proposed park for as low as R50/ha. The costs of land purchase can be mitigated through public-private partnerships or trade-offs with mining interests within the proposed reserve area.

5.8 Conclusions

The systematic conservation planning protocol used in this study has allowed for the identification of a core reserve in the Knersvlakte that achieves an explicit set of conservation targets and reserve design criteria. The proposed reserve presents a conservation vision for the realisation of a major reserve in the Knersvlakte. Implementation of this vision will be gradual, however, the prioritisation exercise has highlighted those areas of the reserve that should form the immediate priorities for action. Also, the implementation of this plan will not be an easy process. Experience over the last seven years in the Knersvlakte and with other conservation projects highlights the need for a dedicated champion that will be able to drive this process within WCNCB and LHSKT as well as with other stateholders. The LHSKT has made funds for land acquisition available and the WCNCB has indicated its willingness to manage the reserve. Combined with this plan, all that remains is for someone to ensure that the implementation process is managed effectively and efficiently to ensure that conservation options are not lost.

5.9 Appendices

Appendix 5.1 A summary of the irreplaceability values and alternative land-use ranks for properties identified for inclusion into the core Knersvlakte Nature Reserve. The numbers in the "Sitekey" column correspond to property labels in Figure 5.26. Site irreplaceability (Irr.) and summed irreplaceability (Summed Irr.) provide a relative rank of the biodiversity importance of each property within the reserve with regard achieving the conservation targets. The initial irreplaceability (Initial Irr.) and summed irreplaceability (Initial Summed Irr.) provide a relative rank of each properties contribution to achieving the conservation targets before any site is reserved. The minerals resource potential and cropping agricultural potential ranks are calculated using the protocol discussed in Section 5.5. The existing "reserved" property, Moedverlooren, is greyed-out in the Table.

Cadastre Name	Sitekey	Area	Irr.	Initial Irr.	Summed Irr.	Initial Summed Irr.	Mineral Resource Potential	Cropping Agricultural Potential
C07800000000017800000 ZOUTFONTEIN	622	2844.05	0.0123	1	0.0123	2.1236	VERY HIGH	HIGH
C07800000000017900000 HOLRIVIER	651	1883.45	0.0168	0.9791	0.0168	0.7473	VERY HIGH	HIGH
C07800000000017800000 ZOUTFONTEIN	641	1339.78	0.0031	0.9067	0.0031	0.6376	VERY HIGH	HIGH
C07800000000018800000 VOGELSTRUIS VLAKTE	704	948.1	0.0214	0.2451	0.0214	0.213	VERY HIGH	HIGH
C07800000000017800004 ZOUTFONTEIN	710	140.57	0.0008	0.0208	0.0008	0.0235	VERY HIGH	HIGH
C07800000000022600000 VARSCHE RIVIER EXTENSION B	606	3166.22	0.0344	1	0.0344	3.3334	VERY HIGH	LOW
C07800000000021500000 QUAGGA'S KOP	544	2632.77	0.0334	1	0.0329	3.0564	VERY HIGH	LOW
C07800000000022500000 ROOIBERG	553	3010.03	0.0347	1	0.0347	2.582	VERY HIGH	LOW
C07800000000021500000 QUAGGA'S KOP	2267	2007.1	0	1	0	2.3696	VERY HIGH	LOW
C07800000000022500000 ROOIBERG	603	1345.26	0.0343	1	0.0343	2.1916	VERY HIGH	LOW
C07800000000021200003 WOLVENEST	548	805.27	0	0.9913	0	1.0073	VERY HIGH	LOW
C07800000000021200002 WOLVENEST	551	128.31	0	0.0788	0	0.0856	VERY HIGH	LOW

Cadastre Name	Sitekey	Area	Irr.	Initial Irr.	Summed Irr.	Initial Summed Irr.	Mineral Resource Potential	Cropping Agricultural Potential
C07800000000021200001 WOLVENEST	552	120.89	0	0.0626	0	0.0739	VERY HIGH	LOW
C07800000000022600000 VARSCHE RIVIER EXTENSION B	626	22.5	0	0.0033	0	0.0052	VERY HIGH	LOW
C07800000000026000004 VARSCHE RIVIER	680	641.39	0.0176	0.8955	0.0175	0.7243	VERY HIGH	MEDIUM
C07800000000021500002 QUAGGA'S KOP	589	235.45	0	0.4726	0	0.3363	VERY HIGH	NONE
C07800000000021500002 QUAGGA'S KOP	585	19.74	0	0.0155	0	0.0216	VERY HIGH	NONE
C07800007000134300000 HOLRIVIER	2343	5611.11	0.9999	1	1.6181	2.9403	HIGH	HIGH
C07800000000017800002 ZOUTFONTEIN	650	2025.81	0.0471	0.9385	0.0471	0.7328	HIGH	HIGH
C07800000000017800003 ZOUTFONTEIN	711	585.94	0.0119	0.1698	0.0119	0.1518	HIGH	HIGH
C07800000000017800008 ZOUTFONTEIN	706	531.41	0.0023	0.1688	0.0023	0.1388	HIGH	HIGH
C07800000000018900002 ZOET VLAKTE	756	81.02	0.0012	0.0082	0.0012	0.0084	HIGH	HIGH
C07800007000031600000 BERGPLAATS	566	7109.07	0.2193	1	0.1265	5.8437	HIGH	LOW
C07800000000020800000 MOEDVERLOREN	565	6792.5	0.0318	1	0.0318	5.7722	HIGH	LOW
C07800000000021300000 QUAGGAKOP	2266	5894.34	0.0016	1	0.0016	4.0871	HIGH	LOW
C07800000000021200000 WOLVENEST	540	3044	0.1038	1	0.1038	3.8459	HIGH	LOW
C07800000000009800000 ZAND KRAAL C	415	6490.1	0.635	1	0.3976	2.6469	HIGH	LOW
C07800000000017800006 ZOUTFONTEIN	681	849.22	0.0003	0.6157	0.0003	0.3458	HIGH	LOW
C07800000000021200004 WOLVENEST	547	149.52	0	0.0199	0	0.0275	HIGH	LOW
C07800000000021500000 QUAGGA'S KOP	560	16.68	0	0.0062	0	0.0089	HIGH	LOW
C07800000000011100003 FLAMINK VLAKTE	414	4835.91	0.086	1	0.086	3.2238	HIGH	NONE
C07800000000011400002 LOT 3475 OOR KRAAL	384	2546.06	0.0024	1	0.0024	2.6684	HIGH	NONE
C07800000000011300000 KARREE BERG	390	4047.82	0.0403	1	0.0424	2.6186	HIGH	NONE
C07800000000012600000 DE TOEKOMST	357	2969.81	0.0041	1	0.0041	2.4699	HIGH	NONE
C07800000000011100001 FLAMINK VLAKTE	472	1351.11	0.0117	0.9574	0.0117	0.7556	HIGH	NONE
C07800000000012400000 SPRINGHAANS KLOOF	347	926.14	0.0011	0.9258	0.0011	0.637	HIGH	NONE

Cadastre Name	Sitekey	Area	Irr.	Initial Irr.	Summed Irr.	Initial Summed Irr.	Mineral Resource Potential	Cropping Agricultural Potential
C07800000000021200005 WOLVENEST	536	876.03	0	0.9117	0	0.6025	HIGH	NONE
C07800000000012200000 ERT VARK GAT	2264	1190.67	0.0016	0.8353	0.0016	0.6021	HIGH	NONE
C07800000000011100002 FLAMINK VLAKTE	438	1796.68	0.0837	0.3328	0.0837	0.2286	HIGH	NONE
C07800000000004900000 HINGS VALLEI	212	3180.54	0.1578	1	0.1578	2.9528	LOW	NONE
C07800000000012400002 SPRINGHAANS KLOOF	327	768	0.0039	0.7181	0.0039	0.3986	LOW	NONE
C07800000000011400003 LOT 3475 OOR KRAAL	373	173.73	0	0.1134	0	0.0968	LOW	NONE
C07800000000020800000 MOEDVERLOREN	637	685.92	0.014	0.1016	0.014	0.1143	MEDIUM	LOW
C07800000000020800001 MOEDVERLOREN	636	19.05	0.0001	0.0015	0.0001	0.0019	MEDIUM	LOW
C07800000000011300001 KARREE BERG	330	4376.71	0.297	1	0.272	3.4056	MEDIUM	NONE
C07800000000011900000 NIEUWOUDTS NAAUWTE	251	2648.16	0.4697	1	0.4697	2.9866	MEDIUM	NONE
C07800000000011500000 SPITS BERG	304	3191.53	0.1997	1	0.19	2.9282	MEDIUM	NONE
C07800000000012000000 DRIE KUIL	301	2437.1	0.125	1	0.125	2.3908	MEDIUM	NONE
C07800000000004800000 KRANTZ KRAAL	236	2570.76	0.0011	1	0.0011	2.1287	MEDIUM	NONE
C07800000000021100000 LUIPERS KOP	513	2151.03	0	0.9996	0	1.8098	MEDIUM	NONE
C07800000000011800000 VINKELS KOLK	249	2589.25	0.2148	0.9989	0.2115	1.4278	MEDIUM	NONE
C07800000000011600000 BIESJES VLEY	293	3526.68	0.3032	0.9925	0.214	1.0218	MEDIUM	NONE
C07800000000011100004 FLAMINK VLAKTE	456	1411.54	0	0.9843	0	0.953	MEDIUM	NONE
C07800000000012100001 ROODE BERG	2262	1056.4	0.0133	0.7652	0.0133	0.4627	MEDIUM	NONE
C07800000000011100005 FLAMINK VLAKTE	481	805.62	0	0.7133	0	0.4231	MEDIUM	NONE
C07800000000012100000 ROODE BERG	303	826.48	0.0269	0.5278	0.0269	0.3144	MEDIUM	NONE
C07800000000021300000 QUAGGAKOP	531	43.3	0	0.0068	0	0.0086	MEDIUM	NONE

Appendix 5.2 A summary of the deeds information for the priority properties indetified in Figure 5.27.

Property Ownership Categories	Owner Name	Area (ha)	Number of Properties	Cadastre Number on Figure 5.27	Each Category as % of Total Area
Mine-owned	1 B P B GYPSUM PTY LTD	805.274	1	2	
	2 CAPE LIME PTY LTD	255.188	2	7, 8	
	3 NATIONAL PORTLAND CEMENT CO LTD	128.311	1	3	
	4 PRETORIA PORTLAND CEMENT CO LTD	762.276	2	4, 17	
Total of mine-owned		**1951.049**	**6**		**4.4**
Privately-owned	1 BEESWATER IND PTY LTD	4184.059	3	11, 13, 14	
	2 BURDEN, ERASMIS JOHANNES	2557.224	2	15, 20	
	3 CLOETE, CHRISTIAAN NATHANIEL	849.219	1	18	
	4 LOUW, CORNELIS PHILIPUS	9083.058	3	10, 12, 24	
	5 M COHEN BOERDERY TRUST	140.573	1	21	
	6 WIESE, TOBIAS GERHARDUS TOBASE	4639.868	2	1, 25	
	7 ZYL, JOHANNES CHRISTIAAN JACOBUS VAN	2550.408	3	16, 22, 23	
	8 ZYL, JOHN WILSON VAN	948.097	1	19	
Total of privately-owned		**24952.506**	**16**		**56.7**
State-owned	1 REPUBLIEK VAN SUID-AFRIKA	17075.462	4	5, 6, 9, 26	
Total of State-owned		**17075.462**	**4**		**38.8**
Total of all categories		**43979.017**	**26**		**100**

Appendix 5.3 Deeds information for priority properties. Property numbers correspond to labels in Figure 5.27.

Property Number	Deeds Number	Deeds Property Name	Owner Name	Owner Category	Owner ID Number	Owner Age	Title Deed Number	Registration Date	Purchase Date	Price Paid for Property	Deeds Information	Survey Information	Area (ha)
1	C07800000000021500000	QUAGGA'S KOP	WIESE TOBIAS GERHARDUS TOBASE	PRIVATE	2109105004003	80	T10128/1966	19660530			YES	YES	2632.769
2	C07800000000021200003	WOLVENEST	B P B GYPSUM PTY LTD	MINE			T75030/1990	19901212	19900405	R14819	YES	YES	805.274
3	C07800000000021200002	WOLVENEST	NATIONAL PORTLAND CEMENT CO LTD	MINE			T10252/1949	19490622			YES	YES	128.311
4	C07800000000021200001	WOLVENEST	PRETORIA PORTLAND CEMENT CO LTD	MINE	189200066706		T36197/1980	19800929			YES	YES	120.886
5	C07800000000022500000	ROOIBERG	REPUBLIEK VAN SUID-AFRIKA	STATE			G29/1944	19440317			YES	YES	3010.031
6	C07800007000031600000	BERGPLAATS	REPUBLIEK VAN SUID-AFRIKA	STATE				19470101			YES	YES	7109.069
7	C07800000000021500002	QUAGGA'S KOP	CAPE LIME PTY LTD	MINE	199900217107		T39367/2000		20000111	R1520291	YES	YES	19.738
8	C07800000000021500002	QUAGGA'S KOP	CAPE LIME PTY LTD	MINE	199900217107		T39367/2000		20000111	R1520291	YES	YES	235.45
9	C07800000000022500000	ROOIBERG	REPUBLIEK VAN SUID-AFRIKA	STATE			G29/1944	19440317			YES	YES	1345.256
10	C07800000000022600000	VARSCHE RIVIER EXTENSION B	LOUW CORNELIS PHILIPUS	PRIVATE	5108275019087	50	T6096/1991	19910204	19900616	R151800	YES	YES	3166.223
11	C07800000000017800000	ZOUTFONTEIN	BEESWATER IND PTY LTD	PRIVATE			T4622/1975	19750303			YES	YES	2844.046
12	C07800000000022600000	VARSCHE RIVIER EXTENSION B	LOUW CORNELIS PHILIPUS	PRIVATE	5108275019087	50	T6096/1991	19910204	19900616	R151800	YES	YES	22.497
13	C07800000000017800000	ZOUTFONTEIN	BEESWATER IND PTY LTD	PRIVATE			T4622/1975	19750303			YES	YES	1339.778
14	C07800000000017800000	ZOUTFONTEIN	BEESWATER IND PTY LTD	PRIVATE			T4622/1975	19750303			YES	YES	0.235
15	C07800000000017800002	ZOUTFONTEIN	BURDEN ERASMIS JOHANNES	PRIVATE	3510045032088	66	T48211/1997	19970527	19961214	R218000	YES	YES	2025.814

Property Number	Deeds Number	Deeds Property Name	Owner Name	Owner Category	Owner ID Number	Owner Age	Title Deed Number	Registration Date	Purchase Date	Price Paid for Property	Deeds Information	Survey Information	Area (ha)
16	C07800000000017900000	HOLRIVIER	ZYL JOHANNES CHRISTIAAN JACOBUS VAN	PRIVATE	3902065012001	62	T55496/1983	19831202			YES	YES	1883.447
17	C07800000000026000004	VARSCHE RIVIER	PRETORIA PORTLAND CEMENT CO LTD	MINE	189200066706		T36197/1980	19800929			YES	YES	641.39
18	C07800000000017800006	ZOUTFONTEIN	CLOETE CHRISTIAAN NATHANIEL	PRIVATE	3403125062086	67	T54525/1993	19930715	19921102	R45000	YES	YES	849.219
19	C07800000000018800000	VOGELSTRUIS VLAKTE	ZYL JOHN WILSON VAN	PRIVATE	3608075011004	65	T47146/1980	19801202			YES	YES	948.097
20	C07800000000017800008	ZOUTFONTEIN	BURDEN ERASMIS JOHANNES	PRIVATE	3510045032088	66					NO	YES	531.41
21	C07800000000017800004	ZOUTFONTEIN	M COHEN BOERDERY TRUST	PRIVATE			T14962/1998	19980223	19970813	R1048922	YES	YES	140.573
22	C07800000000017800003	ZOUTFONTEIN	ZYL JOHANNES CHRISTIAAN JACOBUS VAN	PRIVATE	3902065012001	62	T18813/1958	19581218			YES	YES	585.942
23	C07800000000018900002	ZOET VLAKTE	ZYL JOHANNES CHRISTIAAN JACOBUS VAN	PRIVATE	3902065012001	62	T18813/1958	19581218			YES	YES	81.019
24	C07800000000021300000	QUAGGAKOP	LOUW CORNELIS PHILIPUS	PRIVATE	5108275019087	50	T98746/1997	19971014			YES	YES	5894.338
25	C07800000000021500000	QUAGGA'S KOP	WIESE TOBIAS GERHARDUS TOBASE	PRIVATE	2109105004003	80	T10128/1966	19660530			YES	YES	2007.099
26	C07800007000031600000	HOLRIVIER	REPUBLIEK VAN SUID-AFRIKA	STATE				19970101			YES	YES	5611.106

Appendix 5.4 A summary of the contributors to the vegetation community database compiled by Philip Desmet for Namaqualand

PROJECT CODE	Explanation	Author Code	Number of relevés in dataset
5	Annelise le Roux surveys of Namaqualand		22
6	Annelise le Roux surveys of Namaqualand		6
7	Annelise le Roux surveys of Namaqualand		32
8	Annelise le Roux surveys of Namaqualand		45
9	Annelise le Roux surveys of Namaqualand		45
10	Annelise le Roux surveys of Namaqualand		23
11	Annelise le Roux surveys of Namaqualand		4
12	Annelise le Roux surveys of Namaqualand		2
14	Helga Rosch PhD Goegap Nature Reserve		284
15	Ute Schmiedel PhD data on quartz patches in SA		1593
16	Philip Desmet MSc project on strandveld		118
17	Philip Desmet species list from Kleinzee Nature Reserve		1
18	Philip Desmet survey of DeBeers Buffels River Nuttabooi mining area		9
19	Philip Desmet survey of DeBeers Buffels River Staanhoek mining area		6
20	Tilla Raimondo Honous project survey of Strandveld at Groen River mouth		19
21	Philip Desmet assessment of the Strandveld at Strandfontein		3
22	Philip Desmet assessment of Kookfontein for SANP/WWF		26
23	Philip Desmet Gamsberg EIA		84
24	Tania Anderson Gamsberg EIA		5
25	Philip Desmet IDC Silicone Mine EIA Nuwerus		22
26	Norbert Juergens Namaqualand phytosociological dataset	NJ	1421
27	Bauer data from Muncina	B	14
28	Sue Milton data from Muncina	SM	115
29	Francine Reubin data from Muncina	FR	98
30	B Bayer data from Chris Stokes	BB	1494
31	P Desmet Steytlerville Project for WWF	PGD	76
		Total	5567

Appendix 5.5 Spatial components of ecological processes explicitly or implicitly considered in the reserve design process.

Processes		**Scales Spatial**	**Temporal**	**Human perspective**
1 Abiotic				
1.1 Climate				
1.1.1 Rainfall	• Spatial variation in rainfall	>10 000 ha	seasonal	static?
	• Climate change	>10 000 ha	geological	static?
	• Dispersal (see below)			static
	• Drought (see below)			
1.1.2 Fog	• Temperature moderation and alternative source of moisture for plant growth.	individual plant	daily, seasonal	static
	• Greater diversity of habitats in areas with both high and low lying areas that get fog.	>10 000 ha	evolutionary	static
1.1.3 Wind	• Long distance dispersal.	all levels	daily, seasonal	active
	• Wind erosion of exposed surfaces, e.g. sand fields.	1 ha upwards	seasonal to geological	active
	• Redistribution of plant organic matter influences nutrient cycling, plant recruitment and patchiness.	<0.1 ha	seasonal	active
1.1.4 Temperature	• temperature gradient perpendicular to coast	>10 000 ha	evolutionary/geological	static
1.2 Geological				
1.2.1 Erosion	• Fluvial	<10 000 ha	seasonal to geological	active
	• Aeolean	>10 000 ha	seasonal to geological	active
1.2.2 Tectonics	• Continental uplift	>100 000 ha	geological	static
	• Local tectonics	>10 000 ha	geological	static
1.2.3 Lithology	• Quartz veins a source material for quartz fields.	>1000 ha	geological	fixed
	• Mosaics of different substrata maintain ecological (edaphic) diversification of poorly-dispersed lineages.	1 ha upwards	geological	fixed
	• Distance between rock refugia for bulbs from predation (see below).	max distance related to pollinator type	geological	fixed

	• Grassland vegetation on SAND (Sandveld) is a keystone plant habitat as it is an important summer grazing resource and breeding ground for intra-karoo migratory birds.	and distance pollen moved 10 000 - 100 000 ha	seasonal to geological	active
1.2.4 Pedological	• Soil formation and its contributes to habitat mosaic.	<0.1 ha	geological (but see below)	static?
	• Soil ecological processes as influenced by other processes such as wind erosion, trampling.	<0.1 ha upwards	seasonal to decades	active
1.2.5 Hydrological (geohydrological)	• Saline vs. fresh water rivers as two very distinct riparian habitats for plants.	>100 000 ha	(seasonal) climatic to geological	static (active)
	• Fresh water riverine wash plant communities are important refuge habitat many invertebrates and vertebrates.	<0.1 – 100 ha	geological	static
	• Run-off		seasonal	active
	• Saline seepage areas as important breeding habitat for insects.	<0.1 - 1 ha	seasonal? climatic	active
2 Biotic				
2.1 Dispersal	• Mammal – baboons also important agents of disturbance.	< 10 000 ha	daily to seasonal	active
	• Reptiles (tortoises – although not shown yet is suspected) What is a tortoises home range?		daily to seasonal	active
	• Birds – mostly frugivores and granivores which are probably very nomadic.	>100 000 ha	daily to seasonal	active
	• Water (passive dispersal) movement of seed to inter-plant fields and within community to down-slope within drainage basins.	<0.1 – 10 ha	seasonal to decades	active
	• Wind (passive dispersal) of seed within communities and between rock outcrops and drainage basins.	<0.1 - >1000 ha	seasonal	active
	• Ants	<10 ha	seasonal	active
	• Long-distance by migratory birds and raptors.	>1 000000 ha	seasonal to decades	active
	• Rock outcrops in the sides of incised river valleys provide stepping stones for lithophilus plants.	>10 000 ha	decade to >100 years	active
	• Quartz fields associated with erosional surfaces on the banks of rivers are important stepping stones for plants restricted to quartz patch habitats.	1000 ->10 000 ha	decade to >100 years	active

	• River corridors act as important seed vectors.	>10 000 ha		
2.2 Seed-bank dynamics	• Dispersal (see above)	<100 ha	seasonal	active
	• Longevity	<0.1 ha	seasonal	active
2.3 Pollination This is a very important component of ecological-evolutionary reserve design for plants	• Pollinator biology - where do they eat, what are their food requirements (pollen, nectar, blood), where do they breed? Scale of process dependent on type of pollinator and how feeding habits relate to breeding requirements.	Small distance pollinator (e.g. solitary bees) - <0.1 to 10 ha Long distance (e.g. honey bees) – 10 to >1000 ha	daily to seasonal	active
	• How far does pollen move? (pollinator behavior, very dependent on type of pollinator)	<0.1->1000 ha	daily	active
2.4 Population biology	• Competition	indiv. plant	daily to seasonal	active
	• Facilitation	indiv. plant	daily to seasonal	active
	• Succession	indiv. plant	seasonal to decade	active
	• Recruitment	indiv plant	seasonal to decade	active
	• Death	indiv. plant	seasonal to decade	active
	• Nurse plants	indiv. plant	seasonal to decade	active
	• Patch creation (see below).	indiv. plant to <0.1 ha	seasonal to decade	active
	• Phenology (see pollination).	indiv. plant	seasonal	active
	• Minimum viable population (small, short-lived, high density succulent or geophyte, e.g. *Oophytum oviforme*).	1 ha	seasonal to decade	active
	• Minimum viable population (large, long-lived, low density succulent, e.g. *Euphorbia schoenlandii*).	100 ha	decade	active
2.5 Primary production	• Photosynthesis by higher plant.	indiv. plant	daily to seasonal	active
	• Photosynthesis by biogenic crusts (algal mats).	<0.1 ha	daily to seasonal	active
2.6 Herbivory and the role consumers play in nutrient-cycling	• Episodic herbivore outbreaks (e.g. locusts swarms, springbok migrations)	100 - >100 000 ha	seasonal to decade	active

especially decomposition				
	• Resident invertebrate grazing (from caterpillars to monkey beetle herbivory of flowers).	<0.1 – 10 ha	daily to seasonal	active
	• Rodents (seed collection & herbivory, fossorial rodents) RESIDENT HERBIVORES	<0.1 – 10 ha	daily	active
	• Mammals (e.g. small antelope, sheep, goats and cattle in the absence of large indigenous ungulates). MIGRATORY HERBIVORES	0.1 – 1000 ha	daily to seasonal	active
	• Harvester termites could be the largest consumers of plant biomass in the system - keystone consumers? RESIDENT HERBIVORES	<0.1 – 10 ha	daily	active
2.7 Patch creation (disturbance regimes)	• Living plants	individual plant	decade	active
	• Dead plants	individual plant	decade	active
	• Ants	<0.1 ha	decade	active
	• Harvester termites and the creation of heuweltjie and inter-heuweltjie habitat matrix.	<100 ha	decade to >1000 years	active
	• Burrowing by other arachnids.	<0.1 ha	daily to seasonal	active
	• Rodent foraging creating disturbances and eating bulbs (e.g. porcupines – their digging is probably a keystone process in the life-history or many geophytes).	<0.1 – >1000 ha	daily to seasonal	active
	• Fossorial rodents and other animals turning over soil and creating fields (e.g. golen moles, elephant shrews, whistling rats, ground squirrels).	<0.1 – >1000 ha	decade	active
	• Large burrowing mammals create large disturbances replicated over a much larger areas and require larger areas to maintain minimum viable populations (e.g. aardvarks, bat-eared fox, suricates).	<0.1–>10 000 ha	daily to seasonal	active
	• Large mammal migration and maintenance of viable populations of large ungulates and predators (e.g. rhino and leopard).	>100 000 ha.	decade	static
2.8 Nutrient cycling	• See decomposition by animals, inverts and soil fauna.			
	• Redistribution of nutrients by water and wind – this is a keystone process in arid lands (see Tongway and Reynolds) see above under abiotic section for spatial context.	<0.1 to 100 ha	daily to seasonal	active

2.9 Soil biological processes	• Biogenic crusts	<0.1 ha	daily to seasonal	active
	• Decomposition	<0.1 ha	daily to seasonal	active
	• Symbiotic relationships (mycorrhizal relationships).	<0.1 ha	daily to seasonal	active
2.10 Evolution	• Allopatric speciation where "species" are separated by geographical distance such as different drainage basins or mountain ranges.	>10 000 ha	seasonal to >1000 years	active
	• Sympatric speciation where "species" occupy separate niches across adjacent habitat (e.g. geological or catenal sequences).	<0.1 ha upwards	seasonal to >1000 years	active
	• Whole minor drainage basins associated with quartz fields maintain presumed evolutionary fronts, distinct between basins, consisting of different nested clades of derived taxa	10 –10 000 ha	seasonal to >1000 years	active
	•			

Appendix 5.6 The complete feature table derived for planning the core reserve. 16 features (land-classes) have zero targets. These are classes that are peripheral to the planning domain and occupy very small areas of the planning domain (i.e. <50ha).

	Feature ID	Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
1	KMB	Kamiesberg Mountain Brokenveld	22155.00	21656.50	0.50	11077.50
2	NK	Namaqualand Klipkoppe	211868.25	188356.25	0.50	105934.13
3	PQGP	Platbakkies Quartz and Gravel Patches	386.00	386.00	0.35	135.10
4	BB	Bushmanland Basin	2776.25	2776.25	0.20	555.26
5	NK_F	Namaqualand Klipkoppe Flats	42894.75	41778.50	0.35	15013.16
6	NLSK	Namaqualand Lowland Succulent Karoo	17813.00	16066.75	0.35	6234.55
7	HK	Hantam Karoo	163.75	163.75	0.35	57.31
8	NAG	Namaqualand Arid Grasslands	45694.50	42923.75	0.35	15993.08
9	NKLSK	Northern Knersvlakte Lowland Succulent Karoo	113012.50	112939.75	0.35	39554.38
10	NQSK	Nuwerus Quartzite Succulent Karoo	62456.75	61473.50	0.35	21859.86
11	NA	Namaqualand Alluvia	8006.25	3665.25	0.35	2802.19
12	NRSP	Namaqualand Red Sand Plains	113532.75	91187.75	0.35	39736.46
13	NSF	Namaqualand Sand Fynbos	39252.00	37371.50	0.35	13738.20
14	KNEQP	Knersvlakte Quartzfields	122444.00	121247.00	0.40	48977.60
15	VSR	Vanrhynsdorp Shale Renosterveld	32147.75	31340.50	0.20	6429.56
16	KOTQP	Kotzerus Quartz Patches	112.50	112.50	0.35	39.38
17	KS	Knersvlakte Shales	70623.75	70490.50	0.35	24718.31
18	KOMQP	Komkans Quartz Patches	27330.25	27265.00	0.35	9565.59
19	NSS	Namaqualand Southern Strandveld	1667.25	1626.25	0.35	583.54
20	SKLSK	Southern Knersvlakte Lowland Succulent Karoo	98098.00	81349.50	0.35	34334.30
21	BSF	Bokkeveld Sand Fynbos	52792.25	31736.75	0.20	10558.46
22	CKLSK	Central Knersvlakte Lowland Succulent Karoo	16768.50	16652.25	0.35	5868.98
23	ACSM	Arid Coastal Salt Marshes	3433.50	2627.25	0.35	1201.73
24	RQSK	Rooiberg Quartzite Succulent Karoo	16598.00	16473.75	0.35	5809.30
25	NSG	Namaqualand Spinescent Grasslands	49487.75	47666.25	0.35	17320.71
26	KOEQP	Koekenaap Quartz Patches	1599.25	1549.00	0.40	639.70
27	ORQP	Olifants River Quartz Patches	21402.50	10069.00	0.40	8561.00
28	KD	Knersvlakte Dolorites	2640.25	2558.00	0.40	1056.10
29	TTRQP	Troe-Troe River Quartz Patches	5017.50	5015.00	0.35	1756.13
30	LBF	Lamberts Bay Strandveld	38004.00	29897.25	0.35	13301.40
31	LSF	Leipoldtville Sand Fynbos	51058.00	24006.50	0.20	10211.60
32	DRSK	Doring River Succulent Karoo	11422.25	7472.25	0.35	3997.79
33	RQP	Remhoogte Quartz Patches	3338.75	3135.50	0.35	1168.56
34	ASSK	Agter-Sederberg Succulent Karoo	979.50	438.75	0.35	342.83
35	GSF	Graafwater Sandstone Fynbos	3921.00	2598.00	0.20	784.20
36	WQP	W quartz patches	2263.25	2069.75	0.50	1131.63
37	WIQP	W intermediate heuweltjie/quartz	1252.50	1218.25	0.50	626.25

Feature ID		Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
		veld				
38	NQP	N quartz patches	11383.50	11203.25	0.50	5691.75
39	NIQP	N intermediate heuweltjie/quartz veld	5248.25	5244.75	0.50	2624.13
40	CQP	Central quartz patches	13130.00	13105.50	0.50	6565.00
41	CIQP	Central intermediate heuweltjie/quartz veld	5345.75	5345.75	0.50	2672.88
42	LSTN	Limestone	4920.75	4776.00	0.50	2460.38
43	SEIQP	SE intermediate heuweltjie/quartz veld	757.25	681.50	0.50	378.63
44	SEQP	SE quartz patches	1267.75	1000.00	0.50	633.88
45	SWQP	SW quartz patches	421.75	190.00	0.50	210.88
46	SWIQP	SW intermediate heuweltjie/quartz veld	32.50	3.00	0.50	16.25
47	TYPE_1	Value-1	2.00	2.00	0.33	0.00
48	TYPE_2	Value-2	87.50	74.75	0.33	28.88
49	TYPE_3	Value-3	97499.50	96714.00	0.33	32174.84
50	TYPE_4	Value-4	14649.50	12413.25	0.33	4834.34
51	TYPE_5	Value-5	1039.25	582.00	0.33	342.95
52	TYPE_6	Value-6	15515.00	14424.75	0.33	5119.95
53	TYPE_7	Value-7	95614.50	88284.75	0.33	31552.79
54	TYPE_8	Value-8	18056.00	17133.75	0.33	5958.48
55	TYPE_9	Value-9	159.50	158.50	0.33	52.64
56	TYPE_10	Value-10	932.00	932.00	0.33	307.56
57	TYPE_11	Value-11	14.00	14.00	0.33	0.00
58	TYPE_12	Value-12	2290.00	2064.50	0.33	755.70
59	TYPE_13	Value-13	49369.00	48749.50	0.33	16291.77
60	TYPE_14	Value-14	32.50	32.50	0.33	0.00
61	TYPE_15	Value-15	4.00	4.00	0.33	0.00
62	TYPE_16	Value-16	625.75	617.75	0.33	206.50
63	TYPE_17	Value-17	1484.00	1474.50	0.33	489.72
64	TYPE_18	Value-18	490.00	421.25	0.33	161.70
65	TYPE_19	Value-19	5236.50	5213.50	0.33	1728.05
66	TYPE_20	Value-20	0.75	0.75	0.33	0.00
67	TYPE_21	Value-21	85558.75	81299.50	0.33	28234.39
68	TYPE_22	Value-22	133563.75	92236.00	0.33	44076.04
69	TYPE_23	Value-23	1.00	1.00	0.33	0.00
70	TYPE_24	Value-24	1644.00	1213.00	0.33	542.52
71	TYPE_26	Value-26	5384.75	5381.75	0.33	1776.97
72	TYPE_27	Value-27	24937.00	24509.50	0.33	8229.21
73	TYPE_28	Value-28	351.75	347.25	0.33	116.08
74	TYPE_29	Value-29	45821.50	45276.25	0.33	15121.10
75	TYPE_30	Value-30	36402.75	34560.25	0.33	12012.91
76	TYPE_31	Value-31	65530.75	61744.00	0.33	21625.15
77	TYPE_32	Value-32	76337.50	65960.00	0.33	25191.38
78	TYPE_33	Value-33	726.00	722.00	0.33	239.58

Feature ID		Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
79	TYPE_34	Value-34	20711.50	16343.00	0.33	6834.80
80	TYPE_35	Value-35	175.75	175.75	0.33	58.00
81	TYPE_38	Value-38	1610.00	1544.75	0.33	531.30
82	TYPE_39	Value-39	40798.75	26772.50	0.33	13463.59
83	TYPE_40	Value-40	75770.75	70020.50	0.33	25004.35
84	TYPE_41	Value-41	42.00	42.00	0.33	0.00
85	TYPE_42	Value-42	1806.25	1282.00	0.33	596.06
86	TYPE_43	Value-43	4923.00	4666.50	0.33	1624.59
87	TYPE_44	Value-44	196.25	193.75	0.33	64.76
88	TYPE_45	Value-45	104.25	104.25	0.33	34.40
89	TYPE_46	Value-46	55693.50	45357.50	0.33	18378.86
90	TYPE_47	Value-47	45116.00	41202.75	0.33	14888.28
91	TYPE_48	Value-48	70600.25	63104.50	0.33	23298.08
92	TYPE_49	Value-49	2453.25	2403.75	0.33	809.57
93	TYPE_50	Value-50	20057.25	18954.25	0.33	6618.89
94	TYPE_51	Value-51	35.00	35.00	0.33	0.00
95	TYPE_52	Value-52	25763.75	22707.75	0.33	8502.04
96	TYPE_53	Value-53	4.00	4.00	0.33	0.00
97	TYPE_54	Value-54	10189.25	9354.25	0.33	3362.45
98	TYPE_55	Value-55	1214.50	1132.25	0.33	400.79
99	TYPE_56	Value-56	422.00	422.00	0.33	139.26
100	TYPE_59	Value-59	1537.25	1436.00	0.33	507.29
101	TYPE_61	Value-61	1440.75	1438.25	0.33	475.45
102	TYPE_63	Value-63	87014.25	76741.25	0.33	28714.70
103	TYPE_64	Value-64	9669.50	9374.25	0.33	3190.94
104	TYPE_65	Value-65	64.50	64.50	0.33	21.29
105	TYPE_66	Value-66	14506.00	10084.00	0.33	4786.98
106	TYPE_67	Value-67	570.00	514.50	0.33	188.10
107	TYPE_69	Value-69	28.00	28.00	0.33	0.00
108	TYPE_70	Value-70	46.00	46.00	0.33	0.00
109	TYPE_73	Value-73	6222.75	2407.00	0.33	2053.51
110	TYPE_74	Value-74	688.50	652.50	0.33	227.21
111	TYPE_76	Value-76	7669.00	7462.75	0.33	2530.77
112	TYPE_77	Value-77	564.50	559.25	0.33	186.29
113	TYPE_78	Value-78	308.75	307.75	0.33	101.89
114	TYPE_79	Value-79	770.50	770.50	0.33	254.27
115	TYPE_80	Value-80	7673.00	6349.00	0.33	2532.09
116	TYPE_82	Value-82	7708.00	5708.75	0.33	2543.64
117	TYPE_83	Value-83	4.00	4.00	0.33	0.00
118	TYPE_84	Value-84	7.00	7.00	0.33	0.00
119	TYPE_85	Value-85	109.00	109.00	0.33	35.97
120	TYPE_86	Value-86	4568.25	1991.00	0.33	1507.52
121	TYPE_91	Value-91	57.00	57.00	0.33	18.81
122	TYPE_94	Value-94	7.00	7.00	0.33	0.00
123	TYPE_97	Value-97	1.50	1.50	0.33	0.00

Feature ID		Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
124	TYPE_99	Value-99	1.00	1.00	0.33	0.00

Appendix 5.7 A summary of the estimated amount of each feature transformed and amount currently conserved in all reserves in the planning domain.

Feature name	Original extent of feature in planning domain	Current extent of feature in planning domain	Representation target (% of original extent)	% of feature in planning domain transformed	% of original feature area represented in existing reserves	% of target achieved by existing reserves
SKEP Vegetation Types						
Agter-Sederberg Succulent Karoo	979.5	438.75	35	55.21	0	0
Namaqualand Alluvia	8006.25	3665.25	35	54.22	0	0
Leipoldtville Sand Fynbos	51058	24006.5	20	52.98	0	0
Olifants River Quartz Patches	21402.5	10069	40	52.95	0.76	1.9
Bokkeveld Sand Fynbos	52792.25	31736.75	20	39.88	11.15	55.77
Doring River Succulent Karoo	11422.25	7472.25	35	34.58	0	0
Graafwater Sandstone Fynbos	3921	2598	20	33.74	0	0
Arid Coastal Salt Marshes	3433.5	2627.25	35	23.48	0	0
Lamberts Bay Strandveld	38004	29897.25	35	21.33	6.72	19.19
Namaqualand Red Sand Plains	113532.75	91187.75	35	19.68	0.16	0.46
Southern Knersvlakte Lowland Succulent Karoo	98098	81349.5	35	17.07	0.02	0.05
Namaqualand Klipkoppe	211868.25	188356.25	50	11.1	0.28	0.57
Namaqualand Lowland Succulent Karoo	17813	16066.75	35	9.8	0	0
Remhoogte Quartz Patches	3338.75	3135.5	35	6.09	0	0
Namaqualand Arid Grasslands	45694.5	42923.75	35	6.06	0	0
Namaqualand Sand Fynbos	39252	37371.5	35	4.79	0	0
Namaqualand Spinescent Grasslands	49487.75	47666.25	35	3.68	4.11	11.73
Koekenaap Quartz Patches	1599.25	1549	40	3.14	0	0
Knersvlakte Dolorites	2640.25	2558	40	3.12	0	0
Namaqualand Klipkoppe Flats	42894.75	41778.5	35	2.6	0	0
Vanrhynsdorp Shale Renosterveld	32147.75	31340.5	20	2.51	0.25	1.27
Namaqualand Southern Strandveld	1667.25	1626.25	35	2.46	0	0
Kamiesberg Mountain Brokenveld	22155	21656.5	50	2.25	0	0
Nuwerus Quartzite Succulent Karoo	62456.75	61473.5	35	1.57	0	0
Knersvlakte Quartzfields	122444	121247	40	0.98	4.17	10.42
Rooiberg Quartzite Succulent Karoo	16598	16473.75	35	0.75	7.25	20.72
Central Knersvlakte Lowland Succulent Karoo	16768.5	16652.25	35	0.69	0	0
Komkans Quartz Patches	27330.25	27265	35	0.24	0	0

Knersvlakte Shales	70623.75	70490.5	35	0.19	0	0
Northern Knersvlakte Lowland Succulent Karoo	113012.5	112939.75	35	0.06	0	0
Troe-Troe River Quartz Patches	5017.5	5015	35	0.05	0	0
Bushmanland Basin	2776.25	2776.25	20	0	0	0
Hantam Karoo	163.75	163.75	35	0	0	0
Kotzerus Quartz Patches	112.5	112.5	35	0	0	0
Platbakkies Quartz and Gravel Patches	386	386	35	0	0	0
Focus Habitat Types						
SW intermediate heuweltjie/quartz veld	32.5	3	50	90.77	0	0
SW quartz patches	421.75	190	50	54.95	0.59	1.19
SE quartz patches	1267.75	1000	50	21.12	0	0
SE intermediate heuweltjie/quartz veld	757.25	681.5	50	10	0	0
W quartz patches	2263.25	2069.75	50	8.55	0	0
limestone	4920.75	4776	50	2.94	4.67	9.35
W intermediate heuweltjie/quartz veld	1252.5	1218.25	50	2.73	0	0
N quartz patches	11383.5	11203.25	50	1.58	0	0
Central quartz patches	13130	13105.5	50	0.19	12.15	24.3
N intermediate heuweltjie/quartz veld	5248.25	5244.75	50	0.07	0	0
Central intermediate heuweltjie/quartz veld	5345.75	5345.75	50	0	8.66	17.31
Land-classes						
Value-73	6222.75	2407	33	61.32	8.13	24.63
Value-86	4568.25	1991	33	56.42	9.1	27.56
Value-5	1039.25	582	33	44	11.26	34.12
Value-39	40798.75	26772.5	33	34.38	0.47	1.43
Value-22	133563.75	92236	33	30.94	1.17	3.54
Value-66	14506	10084	33	30.48	1.74	5.27
Value-42	1806.25	1282	33	29.02	0	0
Value-24	1644	1213	33	26.22	0.18	0.55
Value-82	7708	5708.75	33	25.94	6.58	19.95
Value-34	20711.5	16343	33	21.09	0.17	0.5
Value-46	55693.5	45357.5	33	18.56	0.86	2.6
Value-80	7673	6349	33	17.26	1.16	3.51
Value-4	14649.5	12413.25	33	15.27	25.1	76.05
Value-2	87.5	74.75	33	14.57	0	0
Value-18	490	421.25	33	14.03	0	0
Value-32	76337.5	65960	33	13.59	0.64	1.95
Value-52	25763.75	22707.75	33	11.86	0.02	0.05
Value-63	87014.25	76741.25	33	11.81	0.04	0.12
Value-48	70600.25	63104.5	33	10.62	0.37	1.12
Value-12	2290	2064.5	33	9.85	0.02	0.07
Value-67	570	514.5	33	9.74	0	0
Value-47	45116	41202.75	33	8.67	0.2	0.61
Value-54	10189.25	9354.25	33	8.19	0.4	1.21
Value-7	95614.5	88284.75	33	7.67	1.39	4.21
Value-40	75770.75	70020.5	33	7.59	0.65	1.98
Value-6	15515	14424.75	33	7.03	0.13	0.39
Value-55	1214.5	1132.25	33	6.77	4.88	14.78
Value-59	1537.25	1436	33	6.59	0.83	2.51
Value-31	65530.75	61744	33	5.78	0.1	0.32
Value-50	20057.25	18954.25	33	5.5	8.21	24.88
Value-74	688.5	652.5	33	5.23	53.85	163.18
Value-43	4923	4666.5	33	5.21	0	0
Value-8	18056	17133.75	33	5.11	0	0
Value-30	36402.75	34560.25	33	5.06	1.59	4.82

Value-21	85558.75	81299.5	33	4.98	0.02	0.05
Value-38	1610	1544.75	33	4.05	0	0
Value-64	9669.5	9374.25	33	3.05	0	0
Value-76	7669	7462.75	33	2.69	0.28	0.85
Value-49	2453.25	2403.75	33	2.02	3.25	9.85
Value-27	24937	24509.5	33	1.71	0	0
Value-16	625.75	617.75	33	1.28	0.4	1.21
Value-28	351.75	347.25	33	1.28	2.27	6.89
Value-44	196.25	193.75	33	1.27	0	0
Value-13	49369	48749.5	33	1.25	4.54	13.76
Value-29	45821.5	45276.25	33	1.19	4.16	12.6
Value-77	564.5	559.25	33	0.93	0	0
Value-3	97499.5	96714	33	0.81	0	0
Value-17	1484	1474.5	33	0.64	0	0
Value-9	159.5	158.5	33	0.63	0	0
Value-33	726	722	33	0.55	0	0
Value-19	5236.5	5213.5	33	0.44	0.02	0.06
Value-78	308.75	307.75	33	0.32	0	0
Value-61	1440.75	1438.25	33	0.17	2.48	7.52
Value-26	5384.75	5381.75	33	0.06	0	0
Value-1	2	2	0	0	0	0
Value-10	932	932	33	0	0	0
Value-11	14	14	0	0	0	0
Value-14	32.5	32.5	0	0	6.15	0
Value-15	4	4	0	0	0	0
Value-20	0.75	0.75	0	0	0	0
Value-23	1	1	0	0	0	0
Value-35	175.75	175.75	33	0	0	0
Value-41	42	42	0	0	0	0
Value-45	104.25	104.25	33	0	0	0
Value-51	35	35	0	0	0	0
Value-53	4	4	0	0	12.5	0
Value-56	422	422	33	0	0	0
Value-65	64.5	64.5	33	0	0	0
Value-69	28	28	0	0	10.71	0
Value-70	46	46	0	0	0	0
Value-79	770.5	770.5	33	0	0	0
Value-83	4	4	0	0	0	0
Value-84	7	7	0	0	0	0
Value-85	109	109	33	0	8.26	25.02
Value-91	57	57	33	0	0	0
Value-94	7	7	0	0	0	0
Value-97	1.5	1.5	0	0	0	0
Value-99	1	1	0	0	0	0

Appendix 5.8 Palaeo-Relic Plants in Namaqualand.

Namaqualand is littered with plants that are out of place. These are plants that are generally widespread elsewhere in southern Africa in a different climate zone to Namaqualand, but that have disjunct populations in Namaqualand. At some point in the past these plants expanded their distribution into Namaqualand either when the winter rainfall zone shifted north or the summer rainfall boundary shifted south in response to Pleistocene/Holocene climate fluctuations. When the summer-winter rainfall boundary shifted these plants remained locally in sites where they were (a) either able to persist without recruiting (e.g. exceptionally long-lived or resprouting plants), or, (b) specific habitats where viable populations were able to maintain themselves through reproduction and recruitment. The habitats where these species occur are called here refugia.

Studying these plants can provide some indication as to which type of plants are most likely to endure climate change, and more importantly, in which habitats they are likely to do so. Naturally, this has direct practical implications for conservation planning. Targeting these refugia habitats could help buffer a landscape against species extinctions in the face of a changing climate. I have termed these plants "palaeo-relics" as their present distribution in Namaqualand is a function of past climate shifts and not due to present anthropogenic influences.

These plants in the table below can be classified into two groups: (1) winter rainfall zone relics; and, (2) summer rainfall zone relics. Species in each group are relics of Holocene shifts in the boundary between the summer and winter rainfall zones of southern Africa, essentially a shift in the location of the Succulent Karoo as this biome is the ecotone between these two climate zones. The winter rainfall group species have their present centers of distribution in the southern Cape and Little Karoo. The summer rainfall group species have their centers in Namib and the western arid savannas. These groups of species can be expanded to include entire vegetation types. For example, fynbos in Namaqualand can be considered a "relic" vegetation type from when the winter rainfall zone extended much further northwards.

Climate modeling evidence suggests that the fynbos extended into southern Namibia at the height of the last glacial maximum approximately 18 000 ybp (Midgley *et al.* 2001).

Since this time, fynbos has been progressively retreating to its current distribution, and in Namaqualand the vacated landscape has been filled by succulent karoo vegetation. There are also far more winter rainfall relics than summer rainfall relics in Namaqualand. The summer rainfall group comprises long-lived trees that can last much longer without recruiting. It is likely that the summer rainfall incursion pre-dates the last glacial maximum.

There are few Mesembryanthema ("vygies") represented in the list below. These plants probably track the environment too fast. Generation turnover in mesembs is rapid and, with the very rare exception, they are not "persisters" (i.e. long lived plants that rely on infrequent recruitment from seed or resprouting). These plants either move with the changing climate, speciate or go extinct. Their spectacular radiation could also be a recent phenomenon (Desmet *et al.* 1998; Klak *et al.* 2004) related to this most recent climate shift.

An overall conclusion that can be drawn about where palaeo-relic plants occur in Namaqualand is that they tend to be found on upper slopes, especially south slopes, or aquifers of some kind. These landscape features should be targeted as potential refugia for plants in the face of a changing climate.

Questions that arise from these observations:

- Does this pattern of relics exist in the southern Succulent Karoo?
- Is there any archaeological evidence for these shifts?
- How do fossil termitaria (heuweltjies) tie into this story? Is there a link?
- What are the implications for conservation? Which groups of plants will survive a changing climate? How do we design corridors to facilitate this migration in species? How will different functional guilds respond to climate change?

A preliminary list of palaeo-relic plants found in Namaqualand based on personal observations and discussion with other botanists.

Winter rainfall species

Species Name	Location of Disjunct Populations	Habitat in which the population persists and geology	Present Core Area of Distribution	Source of Information
Tylecodon grandiflora	Sandveld west of Kommagas and near Brandsebaai	Dunes	SW Cape and Peninsula	Peter Bruyns Philip Desmet
Ceropegia occidentalis	Strandfontein, Kleinzee, Rosh Pina	Kloof, quartzite	The genus Ceropegia is a southern Cape group	Peter Bruyns
Huernia species	Sparse populations	Various shaded rocky habitats	The genus Huernia is a southern Cape group	Peter Bruyns
Dioscorea elephantipes	Namaqualand hills; Ploegberg, Richtersveld	Upper slopes, gneiss	Little Karoo eastwards	Peter Bruyns Philip Desmet
Euclea undulata	Quartzite hills, northern Knersvlakte	Slopes, quartzite	Little Karoo eastwards	Philip Desmet
Euclea racemosa	Kleinzee-Koingnaas, Port Nolloth, etc	Dune aquifer	Dune thicket to Natal	Philip Desmet
Leucospermum praemorsum	Hondeklip & Koingnaas	Dune aquifer	Bokkeveld, Gifberg, Nardouberge	Generally known; Andrew MacKenzie
Aloe pillansii	Richtersveld	Upper steep slopes, shale and quartzite	Richtersveld	Elsabé Powell
Aloe dichotoma	Brandberg, Sperrgebied and eastern Bushmanland	Upper slopes, various	Namaqualand	Wendy Foden
Aloe microstigma	Rosh Pina; Cornelsberg Bushmanland Inselbergs	South slopes and kloofs, quartzite	Southern succulent karoo and succulent thicket	Philip Desmet Brian Kemble
Willdenowia incurvata	Holgat River and Gemsbok Vlakte	Dune aquifer	SW Cape Sandveld	Philip Desmet
Wiborgia obcordata	Holgat River and Gemsbok Vlakte	Dune aquifer	SW Cape	Philip Desmet
Pachypodium namaquanum	Bushmanland inselbergs, Aggeneys	Upper south slopes, quartzite	Richtersveld	Philip Desmet
Azima tetracantha	Achab, Bushmanland	Kloof, quartzite	Succulent thicket	Philip Desmet

Species Name	Location of Disjunct Populations	Habitat in which the population persists and geology	Present Core Area of Distribution	Source of Information
Crassula arborescens	Rooiberg 2, Knersvlakte	Upper north slope, quartzite	Southern Karoo and succulent thicket	Philip Desmet
Albuca karooica	Rooiberg 2, Knersvlakte Roesyntjieberg, Richtersveld	Upper south slopes, quartzite	Southern Karoo	Philip Desmet
Haemanthus sanguineus	Ploegberg, Richtersveld	Upper south slopes, gneiss	SW Cape fynbos	Philip Desmet
Polemanniopsis marlothii	Ploegberg, Richtersveld	Upper slopes, gneiss	Pakhuis Mountains, SW Cape	Philip Desmet
Delosperma asperulum	Beeswater, Knersvlakte	Slopes, quartzite	SW Cape renosterveld	Philip Desmet
Namaqualand Sandplain Fynbos	Neutral to acid sand pockets in the west coast Sandveld as far north as Koingnass	Dunes	-	-
Kamiesberg Fynbos	Highest peaks of the Kamiesberg mountains	Upper slopes, granite	-	-

Summer rainfall species

Species Name	Location of Disjunct Populations	Habitat in which the population persists and geology	Present Core Area of Distribution	Source of Information
Acacia erioloba	Kommagas	Dune aquifer	Kalahari eastwards	Philip Desmet
Capparis hereroensis	Bitter River dunes	Dune aquifer	Central Namib Desert	Ernst van Jaarsveld
Namaqualand tree flora in general	Namaqualand	North-facing slopes or riparian habitats	Palaeotropical, summer rainfall genera	

6 Designing a Biosphere Reserve for the Knersvlakte

6.1 Introduction

The previous chapter looked at the design of a core reserve in the Knersvlakte that focussed on representing some of the unique biodiversity pattern features of the region within a statutory reserve. This chapter focuses on planning for large-scale ecological processes that due to the spatial scales over which they operate, cannot be entirely included within a single statutory reserve, i.e. a reserve less than 100 000 ha in extent. Accommodating such processes requires a landscape or regional level approach to conservation planning. The biosphere reserve model used in this study is a convenient conceptual model for integrating an expansive array of biodiversity considerations into regional land-use planning. This chapter develops a design for the Knersvlakte biosphere reserve based on an explicit set of targets and design rules to capture specifically ecological processes, but also biodiversity pattern.

Conservation cannot happen in isolation from humans (Batisse 1996; Abbitt 2000; Margules and Pressey 2000a; Anon. 2002; Young and Fowkes 2003). It needs to consider the whole landscape, natural and anthropogenic, and involve stakeholders from all sectors. The biosphere reserve is one model with which conservation and human development needs can be integrated into a single regional spatial development plan. The focus in this chapter has been broadened from identifying a single core reserve in the Knersvlakte to consider this reserve in the context of the whole landscape, i.e. broadly the Knersvlakte bioregion. This was at the request of Western Cape Nature Conservation Board (WCNCB). The pages that follow provide a brief background on what a biosphere reserve is and also how this has been applied to the Knersvlakte.

The Provincial Government of the Western Cape has since October 2000 advocated a bioregional planning approach, which is intended to give practical effect to South Africa's obligations regarding a number of international agreements including Agenda 21, the Convention on Biological Diversity and the New Partnership for Africa's Development (Anon 2003a). The foundation of bioregional planning is to promote sustainable development that is "development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs" (Brundtland Commission Report 1987 in Anon 2003a). Biosphere reserves have been proposed as the model with which land-use planning can integrate the bioregional planning philosophy into local land-use planning in the Western Cape Province (Anon 2003a). For this reason this study has chosen to use the biosphere concept and terminology when developing the regional conservation vision to facilitate the ease with which these findings are integrated with other land-use planning products.

Biosphere reserves are protected areas included in a global network organized by the United Nations Educational, Scientific and Cultural Organization (UNCESCO). They form part of UNESCO's Man and the Biosphere Program, a global research effort dealing with people-environment interactions over the entire realm of bioclimatic and geographical situations of the biosphere (UNESCO 1996). They are a land-use planning/conservation model that attempts to combine conservation and sustainable development (Phillips 1995). The efficacy of biosphere reserves as vehicles for biodiversity conservation is debatable, however, as a concept for regional land-use planning the model holds much potential.

All biosphere reserves should have the following characteristics (Batisse 1982):

- They are protected areas of land and coast environment;
- Together they constitute a worldwide network linked by a common understanding of purpose, standards and exchange of scientific information;
- Each reserve should include one of more of the following: representative examples of natural biomes; unique communities or areas of unusual features of exceptional interest; examples of harmonious landscapes resulting from traditional patterns of land-use; and/or examples of modified or degraded ecosystems that are capable of being restored to more-or-less natural conditions;
- Each reserve should be large enough to be an effective conservation unit and to accommodate different land-uses without conflict;
- Should provide opportunities for research, education and training, and have value as particular benchmarks or standards for measurement of long-term changes in the global biosphere as a whole;
- The reserve must have adequate long-term legal protection; and,
- In some cases the reserve can coincide with, or include existing or proposed protected areas such as national parks, nature reserves or conservancies.

Biosphere reserves were originally envisioned to contain three physical elements (sometimes referred to as the fried egg concept): one or more core areas, which are securely protected areas conserving biodiversity; well defined buffer areas, usually surrounding or adjoining the core; and, transitional areas that may contain a number of human land-uses (Phillips 1995). Interpretation of the extent and arrangement of these areas is dependent on local conditions and circumstances.

The rationale for developing a biosphere reserve should also have the following broad goals in mind (Phillips 1995):

- Use the reserve to conserve natural and cultural diversity;
- Use the reserve as a model of land management and approaches to sustainable development; and,
- Use the reserve for research, monitoring, education and training.

These goals encompass the envisioned role of the biosphere reserve in the context of the Knersvlakte, i.e. a system that is a vehicle for bio-centric, landscape-wide land-use planning that can assist in the implementation of the principles of sustainable development, rather than a strict conservation tool.

This bio-centric view of land-use planning converges with that of current conservation thinking. The focus of contemporary conservation planning is moving away from individual species and reserves to consider biodiversity and ecological processes at the whole landscape level (Cowling *et al.* 1999a; Dale *et al.* 2000; Clout 2001; Boutin and Hebert 2002; Leitao and Ahern 2002; Pressey *et al.* 2003a; Rouget *et al.* 2003a). The structure of natural landscapes and connectivity within and between areas and regions are central tenets of this philosophy. Thus, the convergence between land-use planning and conservation requirements under the umbrella of bioregional planning make the application of the biosphere reserve model an ideal development framework for the Knersvlakte.

Since the conservation goal of the biosphere reserve is centered on the persistence, connectivity and movement of biodiversity within the whole and adjacent landscapes, it is appropriate to define the boundaries of the Knersvlakte Biosphere Reserve as being the boundaries of the planning domain. The natural environment covers the entire landscape and as such land-use planning should explicitly consider biodiversity in all parts of this landscape not just in pristine areas or reserves. The central theme of the Knersvlakte biosphere reserve should be one of biological linkages - linkages between the escarpment and coastal areas; and, Namaqualand and the Cederberg. This overarching theme of connectivity and maintenance of ecological processes is discussed in the ecological process Chapter 5.2.2) and targets (Chapter 4) sections.

Planning for the biosphere reserve in this project considers only those land-use planning categories that are broadly compatible with biodiversity conservation and where biodiversity conservation targets can be achieved (Table 6.1). Any land-use activities that do not result in medium to long-term degradation or transformation of natural habitats can contribute to meeting conservation targets, hence the inclusion of category Ca, extensive agricultural areas (Table 6.1). These areas are used predominately for livestock grazing. Well-managed small-stock grazing is potentially compatible with and even an essential process for the persistence of biodiversity in Succulent Karoo rangelands (Palmer *et al.* 1999). Areas maintained as natural rangeland and not converted to pastures or other forms of intensive agriculture (e.g. cropping, ostrich farming) can contribute to meeting biodiversity conservation targets.

Table 6.1 Summary of biosphere reserve land-use planning categories (from Anon 2003a)

Category	Category Description
Biodiversity compatible categories considered in this project:	
A Core Areas	
Aa	Wilderness areas
Ab	Other statutory conservation areas
B Buffer Areas	
Ba	Public conservation areas
Bb	Private conservation areas
Bc	Ecological corridors
Bd	Rehabilitation areas
C Agricultural Areas (also Buffer)	
Ca	Extensive agricultural areas
Biodiversity-incompatible categories not considered in the project:	
C Agricultural Areas	
Cb	Intensive agricultural areas
D Urban-Related Areas	
E Industrial Areas	
F Surface Infrastructure and Buildings	

The reserve identified in the previous Chapter will be a statutory reserve and should be designated as a core area (Category A) in the biosphere reserve. There are, however, many other areas within the planning domain that warrant being designated as core areas either as statutory or non-statutory reserves. All biodiversity pattern in the planning domain cannot be represented within a single reserve. However, the location of other potential core areas that achieve biodiversity pattern targets is not addressed in this study. This is an important point to remember when interpreting the results of this study. The formulation of the landscape functionality process targets used to design the biosphere reserve is such that they subsume pattern targets for biodiversity pattern. Therefore other core areas would be located somewhere within the areas identified here under the biodiversity compatible land-use zones. It must be borne in mind that the product of this project is an outcome that will need periodic revision and refinement as the biosphere reserve in the region is implemented and new information comes to hand.

The steps involved in designing the biosphere reserve are similar to those used in designing the core reserve and are not repeated here. The biodiversity features used and alternative land-use options for the planning domain were discussed in the previous chapter. The sections that follow discuss the biosphere reserve goals, the development of targets for ecological processes and the actual design process.

6.2 Biosphere Reserve Goals

From a conservation perspective, the primary goal of the biosphere reserve is to ensure the persistence of ecological processes, in other words retain the natural fabric of the landscape. As many ecological processes cover large areas, the biosphere reserve forms an integral part of retaining the ecological functionality both of the core reserves as well as the landscape as a whole. Thus, it could be said that the primary goal of the biosphere reserve is to maintain a "living landscape". A second goal of the biosphere reserve is to have representative examples of all mapped biodiversity pattern features within the biodiversity compatible land-use zones of the reserve.

The biosphere reserve is not strictly a conservation vehicle. It is mechanism that integrates land-use and conservation needs into a single planning framework. The aim of the bioregional planning philosophy is to promote sustainable development through combining these traditionally separate planning frameworks (Anon 2003a). Thus, the primary goal can be refined to say that the purpose of this study is to identify areas in the landscape that are required to maintain the ecological integrity of the landscape and the products are intended for both land-use planners and conservation planners. In line with the sustainable development drive towards a convergence of human, economic and environmental planning into a single bioregional planning approach, the goal with the outputs of this study is to produce a single set of products whereby different practitioners can interpret the same information differently depending on their context. This need to simplify academically rigorous research outputs into understandable, easy to use products for application by planners is a challenge that currently faces conservation planning generally (Driver *et al.* 2003a).

As this project is an initial attempt at broadly defining the layout of the biosphere reserve, no attempt is made to designate zones within the biosphere reserve beyond the distinction of biodiversity compatible zones (biosphere categories A, B and Ca) and non-compatible zones (biosphere categories Cb, D, E and F). The assumption is made that any area in categories A to Ca, whether designated statutory reserve, conservancy or stock farm, can contribute towards achieving the process targets identified.

In the context of this project, the biosphere reserve is aimed at achieving primarily ecological process targets as these have greater spatial requirements than biodiversity pattern targets (see Section 6.3). Although the biodiversity compatible land-use zones of the biosphere reserve do achieve the targets for the representation of biodiversity pattern and small-scale processes, for implementation these targets should ideally be achieved within a statutory and non-statutory reserve network. The goal of this project is not to identify a network of core reserves in the planning domain that can achieve all patterns targets. There will be numerous other areas within the planning domain that warrant being formally conserved, i.e. biosphere categories A or Ba. As was discussed in the previous section, these areas will need to be identified if the goal is to conserve a representative sample of each biodiversity feature in at least one statutory reserve. Identification of these areas will require further analyses of this dataset to identify exactly where these areas should be located. These areas, however, will all fall within the biodiversity compatible land-use zone identified here.

6.3 Biosphere Reserve Targets

The primary goal of the biosphere reserve in this study is the preservation of landscape ecological functionality through conservation of ecological processes. For the biodiversity represented in the core reserve considered in the previous chapter as well as in the wider landscape to persist, ecological processes need to be explicitly considered in the planning process (Franklin 1993; Noss 1996b; Balmford *et al.* 1998; Cowling *et al.* 1999a; Margules and Pressey 2000a). Just like humans, all other organisms on this planet require space to be able to move, breath, feed and reproduce. This dynamic can collectively be included in the term "ecological processes". In this study, estimating how much space organisms require to survive and persist is the ecological basis for the formulation of process targets.

The core reserve effectively represents many of the globally unique biodiversity attributes of the Knersvlakte. For this biodiversity to persist, one needs to consider landscape-scale ecological processes that allow biodiversity to persist indefinitely. These larger-scale processes invariably revolve around (1) allowing organisms to move freely through the landscape; (2) maintaining environmental features that promote evolutionary processes; and, (3) maintaining environmental features that allow organisms to persist locally in the landscape in the face of a changing environment. Conservation of these processes is not only important for the biodiversity of the core reserve but more importantly for all biodiversity in the landscape.

Experience in South Africa has shown that conservation of processes is "land hungry" requiring that areas outside of formal protected areas be considered in the broader conservation equation (Desmet *et al.* 1999; Cowling *et al.* 2003b). Since preservation of ecological processes requires a larger view of the landscape than that afforded by the formal reserve network, the biosphere reserve model is useful in this context as it addresses land-use across the entire landscape.

The systematic conservation planning approach requires that all biodiversity features be mapped spatially in order for them to be targeted. Biological information such as vegetation type maps or species locality data are useful surrogates for biodiversity pattern. Spatial data on ecological processes are not widely available; therefore, incorporating processes into conservation plans requires a more creative use of available spatial data in order to represent processes. This is one of the biggest challenges facing conservation planning at present.

Processes can be incorporated into the planning process via three major routes (Pressey *et al.* 2003b):

- As explicitly mapped process features that are fixed in space (e.g. riparian corridors or edaphic interfaces); and,
- As proportions of biodiversity pattern features that act as surrogates for processes but which are not fixed spatially, i.e. it can be any proportion of a pattern feature (e.g. minimum area of vegetation type A required to maintain a viable population of species B).
- Design rules that specify spatial arrangement of landscape to achieve process targets.

Processes that are linked to specific areas of the landscape, such as riparian corridors or edaphic interfaces, are relatively easy to incorporate into the planning process. These processes are fixed in space and relate to specific defined landscape features. These processes can be hard-wired into the planning process by mapping and setting targets for these landscape features. Thus, it is assumed that the processes associated with them will be effectively considered in the planning process.

Other processes are much less specific about where in the landscape they operate. These include migration or upland-lowland movement corridors, or minimum areas required to maintain viable populations of organisms. Generally, these require large tracks of natural landscapes to operate, however, they are not linked to specific landscape features. Rather these processes could be linked to part or all of a variety landscape or biodiversity features. Targeting such processes in conservation planning is much more difficult especially where there are choices as to where to locate areas to capture these processes.

One approach used in South Africa has been to physically map upland-lowland gradients and then use these areas as hard-wired features in the planning process (Rouget *et al.* 2003b). This approach is really only appropriate in highly transformed landscapes where there are generally very few options for creating such linkages in the landscape, i.e. the linkages can only be conserved in the areas where they are mapped. In a landscape such as the Knersvlakte where the levels of transformation are generally very low, there are many options as to where to create such linkages. By fixing the location of linkages to specific areas before beginning with the planning process, the outcome of the planning process is effectively constrained before options for all features have been reviewed (Figure 6.1). This results in a Catch 22 situation. By not explicitly mapping the location of linkages allows for flexibility as to where they are created, however, this also creates the situation whereby linkages have to be incorporated into the design process via non-hardwired process targets and/or a set of design rules.

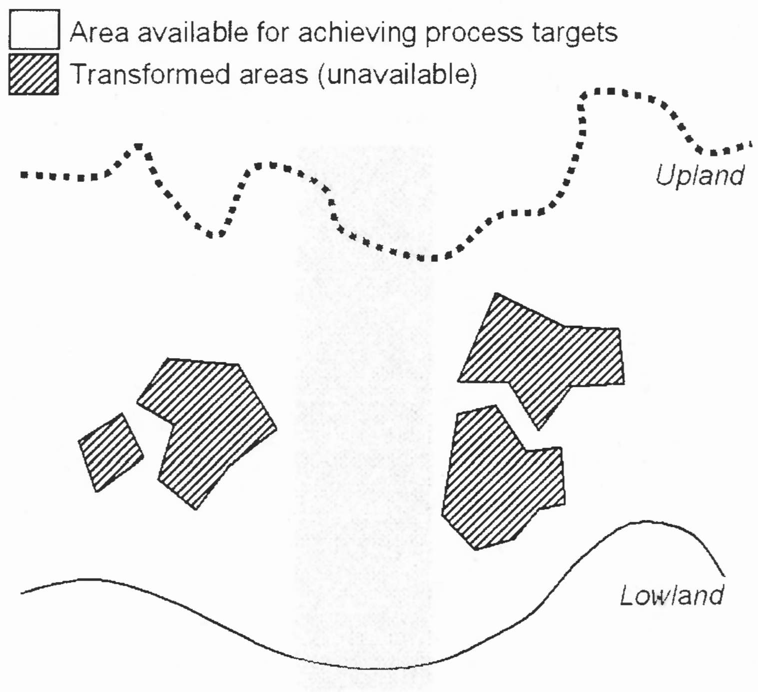


Figure 6.1 A conceptual model for targeting upland-lowland gradients. The shaded area represents a fixed corridor linking the uplands to the lowlands, however, there is flexibility in this landscape as to where this corridor is located so long as it avoids transformed areas. By fixing this corridor to this location as a hard-wired mapped process feature before beginning with the planning process effectively rules out all other options for where this corridor could be located.

The disadvantages of this approach are that it requires informed ecological knowledge of the landscape to design the biosphere; and, any GAP analysis or conservation performance measure cannot determine how well these processes are conserved, as they are not explicitly mapped or targeted. A more detailed study of the biosphere reserve design should investigate "hard-wiring" these processes into the feature set. No solution to this problem is presented in this study. For designing the biosphere reserve a combination of hard-wired, "non-hard-wired" and rule-based process targets are used. Thus, in this project the "expert" plays a critical role in the design of the biosphere reserve.

When considering which processes to include in the planning dataset, the central themes linking the set of processes considered revolves around maintaining landscape functionality (i.e. enough area for biodiversity to persist in place) and connectivity (i.e. linking areas to allow biodiversity to move in space).

6.3.1 Hard-wired processes

Three processes related features were mapped in the landscape – edaphic interfaces, riparian corridors and topographic climate refugia (see Chapter 5.2.2). These discrete landscape features act as spatial surrogates for ecological processes and are fixed in space and therefore can be clearly mapped or hard-wired into the planning process, i.e. there is a unique feature GIS layer in the feature dataset for each of the three process surrogates.

The ecological processes associated with these features are considered as important and therefore all or most of each feature is targeted (Table 6.2). Experience has shown that setting process targets to 100% complicates the design process as sites with tiny areas of processes features present become totally irreplaceable even if there are no other features present at the site (Desmet *et al.* 1999). By setting a process target to less than 100% means that such sites do not over influence decisions, especially since such slivers of process features may be within the boundary error of the process GIS layer and could potentially not even actually be represented at a site. Beyond this reasoning there is no biological rationale for the targets set for these features except that they are important and ideally as much of each feature should be conserved as possible. The targets for topographic climate refugia also reflect a perceived scale of importance in the contribution of each feature towards ecological processes (Table 6.2).

Table 6.2 Process targets for hard-wired ecological process features expressed at percentage of original extent.

Process	Feature Name	% Target
Topographic climate refugia	Sea-facing topographic climate refugia	90
	South-facing slopes climate refugia	80
	Topographic climate refugia	70
Edaphic Interfaces	Edapic interfaces (buffer 500m)	80
Riparian Corridors	River Order 1 (buffer 50m)	80
	River Order 2 (buffer 100m)	80
	River Order 3 (buffer 200m)	80
	River Order 4 (buffer 250m)	80
	River Order 5 (buffer 500m)	80

6.3.2 "Non-hard-wired" processes

The species-area relationship was used to set biodiversity pattern targets to estimate how much area is required to represent a given percentage of the regions flora (Chapter 3). Unfortunately, there are no such empirical methods for setting process targets. For some explicitly mapped process surrogates that are fixed in space such as riparian corridors or edaphic interfaces the importance of the processes associated with these landscape features is such that all of these features are targeted (see Section 6.3.1). For processes that require space but that cannot be attached directly to any specific landscape feature, such as the area necessary to maintain a viable population or to maintain an upland-lowland linkage, the target will not be 100 percent of a feature but rather a proportion of one or many other biodiversity pattern or process features. For example, dune molerats may require X ha of vegetation type A in order to maintain a viable population, but this area can be located anywhere within the rage of the vegetation type A.

Chapter 4 demonstrates that there is a substantial body of research into fragmented landscapes and meta-population dynamics that can provide guidance when setting process targets for landscapes. This research is useful in that many studies examine the biological repercussions (e.g. species loss or reproductive success) of habitat loss (e.g. deforestation, ploughing or urbanisation) in a landscape. In other words, these studies looked at the impact of habitat loss of ecological processes. This research can also be interpreted in terms of how much area is required to maintain ecological processes. The research reviewed in Chapter 4 suggests that ecological processes begin to noticeably break down when landscapes are more than 40% transformed. This is an average across a range of organisms, landscapes and studies.

This threshold for ecological processes can be interpreted as a generic landscape target for processes, i.e. maintain 60% of the landscape in order to maintain minimum of processes. Rather than applying this target to an undefined landscape, this 60% can be distributed through the landscape by applying it to vegetation types and land-classes. Therefore, in this study a landscape functionality process target of 60% of original extent of each vegetation type and land-class was set to accommodate "non-hard-wired" ecological processes.

What this target means is that any 60% of the landscape, in this case it is being applied to individual vegetation types and land-classes, is required to maintain a minimum of ecological process to all the majority of biodiversity to persist in that landscape. The target applies to a biodiversity pattern feature (i.e. vegetation type of land-class), but is not specific about which 60% of that feature is required to meet the target. There is flexibility as to where this target is met within a feature provided the current extent of the feature is greater than 60% of it's original extent. Deciding on where to achieve this target is facilitated partly by the location of other features such as the hard-wired processes and other biodiversity pattern features (e.g. SKEP expert mapped areas); and, partly by a set of design-rules aimed at promoting conceptual requirements for ecological processes.

6.3.3 Design rules to accommodate processes

One of the central tenets of the biosphere reserve is maintaining connectivity of the landscape. Conceptually, the ultimate biosphere reserve plan of biodiversity compatible land-use zones should resemble the connected landscape illustrated in Figure 6.2. To achieve this connectivity in the design process a set of design rules that specifically target three spatial scales of landscape-level connectivity or processes is introduced.

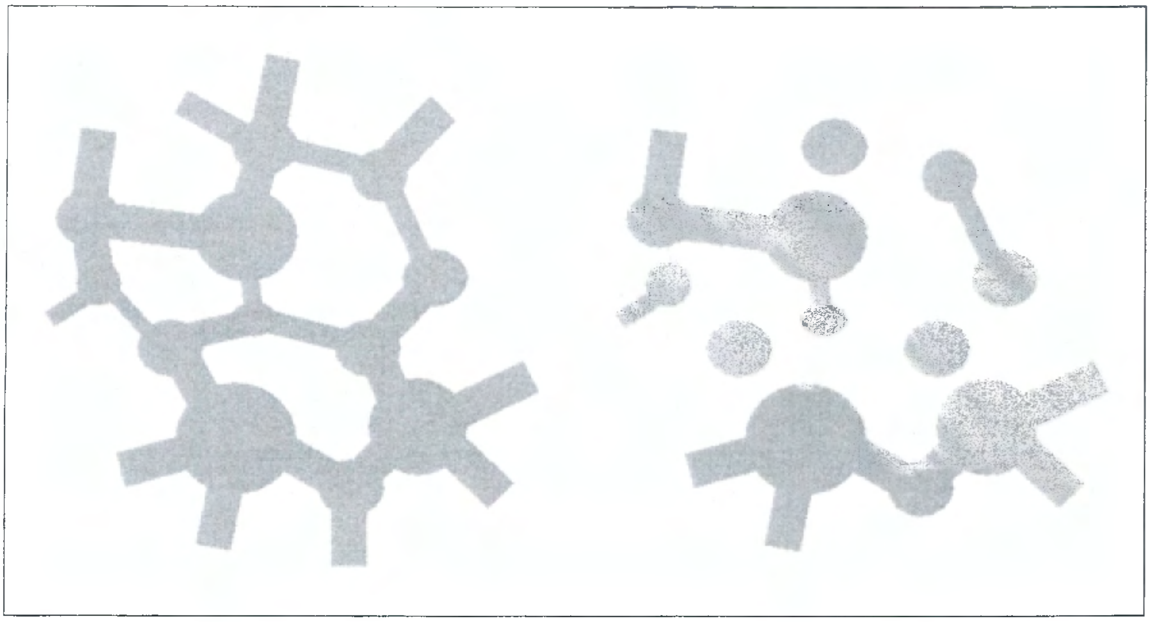


Figure 6.2: A conceptual vision for landscape connectivity to be achieved by the biosphere reserve. Where possible, the biosphere reserve should strive to maintain a more connected landscape such as that on the left rather than the disconnected one on the right. Where a landscape consists only of isolated fragments such as than on the right, the biosphere reserve should strive to restore connectivity in the landscape through appropriate land-use planning.

In addition to other process and pattern features represented in the planning dataset, design rules are also required to give effect to the landscape functionality process targets. These design rules target three classes of landscape-scale ecological processes that cannot be hard-wired but that need to be considered conceptually in the design of the biosphere reserve (Figure 6.3). These are, in decreasing order of spatial requirements:

- Climatic latitudinal gradients are broadly orientated parallel and perpendicular to the coast. These gradients follow the major axes of climatic variation in southern Namaqualand. These are predicted to be the dominant axes of species migration in the face of a changing climate (Midgley *et al.* 2001). With a predicted hotter and dryer climate, Succulent Karoo species are expected to migrate southwards tracking the changing climate.
- Lowland-lowland connectivity corridors link lowland areas in different regions of the Succulent Karoo. These corridors allow lowland species to migrate between regions relatively unhindered by unsuitable mountainous habitats. There are two such corridors in the planning domain: (1) The Doring River and Koebee "Gap"

linking the Knersvlakte with the Tankwa Karoo; and, (2) Krom River and Kliprand "Gap" linking the Knersvlakte with southern Bushmanland.

- Lowland-upland altitudinal gradients range from short gradients between a valley and the crest of a hill to large-scale gradients from, for example, the coast to the top of the Kamiesberg. Altitudinal change allows species to moderate their local environment over small spatial scales simply by migrating vertically. In the face of a changing climate preserving altitudinal gradients will be the most effective means of buffering the landscape against the extinction of species due to a hotter and dryer climate.

The design rules make explicit the need to consider landscape connectivity at a variety of scales. The drawback associated with using rules to achieve this is that it is very difficult to quantitatively measure how well the rules or connectivity targets have been adhered to during the planning process. Post hoc analyses using landscape structure metrics could be used to calculate the degree of landscape connectedness, but this is a tedious process especially where many reserve configurations are possible. Incorporating software such as FRAGSTATS (Li *et al.* 2001; McGarigal 2002) into C-Plan could prove to be useful for measuring and comparing the landscape metrics of any potential reserve network.

6.3.4 Biodiversity pattern

By default the landscape functionality targets include the pattern targets for vegetation types and land-classes. Therefore it is not necessary to set separate pattern targets for these features. Achieving the process targets for these features will also achieve their pattern targets. The SKEP expert mapped areas were included in the feature dataset and the SKEP targets for these were used here (Driver *et al.* 2003b).

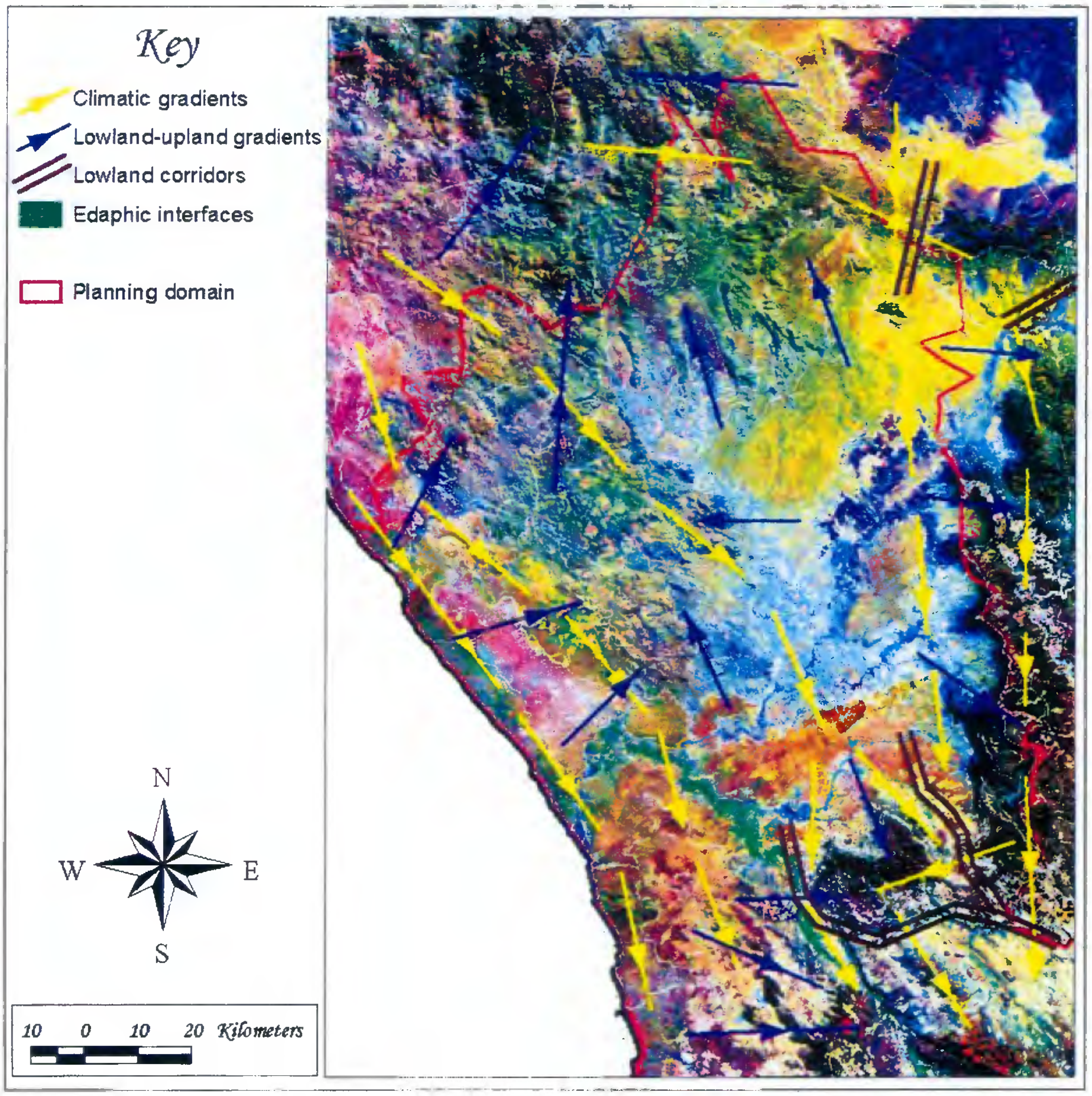


Figure 6.3: A schematic illustration of the landscape-scale ecological processes incorporated into the biosphere reserve design process via means of explicit design rules.

6.3.5 Considering Landscape Aesthetics

The role of landscape aesthetics in the biosphere reserve has been discussed in Chapter 5.2.3. Ideally it would be preferable to protect the entire viewshed of the core reserve, i.e. target of 100% of the viewshed. The viewshed although included in the feature dataset was excluded from the calculation of irreplaceability values because it is not a strict surrogate for biodiversity pattern or process. Including it would bias the reserve design, as irreplaceability values would not reflect true biodiversity-based conservation priorities.

Table 6.3 Patterns targets for SKEP expert mapped areas expressed as percentage of original area.

Feature Group	Feature Name	% Target	Feature Group	Feature Name	% Target
Plants	Plant pgd41	100	Plants cont.	Plant ms4	75
	Plant pgd18	75		Plant ms6	100
	Plant pgd14	10	Invertebrates	Invert JI32 Marble hills	100
	Plant pgd22	12		Invert JI33 Kommandokraal	100
	Plant pgd20	12	Fish	Fish f2	50
	Plant pgd15	12		Fish f4	100
	Plant pgd19	12		Fish f3	50
	Plant pgd17	50	Birds	Birds b22	50
	Plant pgd16	50	Amphibians	Amphibian ao7	10
	Plant ms3	50		Amphibian ao6	10
	Plant ms5	75			

The utility of the viewshed is that where trade-offs are possible between which sites to select to achieve targets, those in the viewshed are selected over those not in the viewshed. The viewshed is also useful for helping to delineate where corridors should be located. When there are choices as to where to locate corridors, these should ideally be located within the viewshed.

The rapidly developing southern viewshed of the core reserve is that most likely to be transformed in the short to medium term (Chapter 5.5). The biosphere reserve design focuses on maintaining this viewshed through manually selecting areas that fall within this viewshed.

6.3.6 Considering Competing Land-Uses

Competing land-uses in the planning domain have been discussed in Chapter 5.5. At the scale of this study urban and industrial expansion are not considered, although they should be considered once this plan gets refined to a relevant scale.

The most extensive landscape transformation initiatives currently in operation in the planning domain are:

1. The continuing expansion of the lower Olifants River irrigation cropping agriculture region.
2. The proposed expansion of this irrigation area as part of the wider Olifants-Doring irrigation scheme (WODRIS).
3. The expanding mining industry along the coast (diamonds and heavy minerals) and in the central Knersvlakte (limestone, diamonds and silica).

These alternative land-use activities have been taken into account during the design of the biosphere reserve by using the land-use potential maps as an information backdrop during the design process. C-Plan does have the capacity to incorporate alternative land-use data into the planning dataset to help visualize the trade-off between biodiversity and alternative land-use development goals (Pressey *et al.* 1995b; Anon. 2001). As the alternative land-use information used in this study is qualitative activity potential maps it was not possible to set quantitative targets for alternative land-uses with which to make effective trade-offs, therefore, it was not incorporated into the C-Plan dataset. Also, it was felt after constructing a trial dataset using the WODRIS data that incorporating this information, either by setting targets for agricultural development or simply as a biodiversity feature vulnerability index, it did not contribute to the design process more that what could be achieved through visual inspection of the data. Unfortunately C-Plan does not have the ability to include site vulnerability information. The only way to incorporate qualitative vulnerability information directly into the planning datasets is if it is associated with a particular biodiversity feature, e.g. a vegetation type. In many cases alternative land-uses target sites irrespective of the biodiversity features present such as with mineral resources or land close to water for irrigation. This limitation is something that should be addressed in future versions of C-Plan.

Overall areas of potential conflict have been avoided where alternative options for achieving conservations are available.

6.4 Designing the Biosphere

Designing the biosphere reserve follows a similar protocol to that adopted in the design of the core reserve. The difference here is that the whole planning domain is considered; the planning units are grid-cells and not cadastres; the feature-set is larger and includes process-related features; and, with the exception of the viewshed all features are targeted.

The viewshed is not included in the initial feature-set, as it is not a biodiversity feature. Constraining options for achieving biodiversity targets by the imposition of an essentially cultural feature detracts from being able to view where the best options for achieving biodiversity targets lie. The viewshed is included later in the planning process during the design phase to guide the selection of areas to achieve both biodiversity and the viewshed targets.

As explained in Chapter 5.2.5, the planning units were changed to reflect the specific goals of the biosphere planning process. The goal is to identify the potential layout for the reserve and not identify individual parcels of land for inclusion into a reserve. In the biosphere reserve, an individual parcel of land can be zoned as many different categories depending on existing land-use and degree of transformation of the cadastre. It is therefore impractical and inefficient to use whole cadastres as planning units where only a part of a cadastre could be required to achieve any given biodiversity target.

Six basic steps were involved in designing the biosphere reserve. These were:

1. Lay out options for all features (excluding the viewshed).
2. Use a minimum set to select sites that meet all targets.
3. Apply the design criteria to the design of the reserve by swapping selected sites with alternative sites that meet both the targets and design criteria. This step uses the viewshed to guide the manual selection of areas.
4. Use a minimum-set to remove any redundant sites from the selected areas, i.e. sites that do not contribute to meeting targets.
5. Repeat steps three and four till all targets and design criteria are satisfied.
6. Manually convert grid cells to polygons relating boundaries to actual mapped features.

6.4.1 Biosphere Design Step 1: Look at options.

The irreplaceability map in Figure 6.4 is a map of options for achieving biodiversity targets. Given the large number features in the dataset the analysis used summed irreplaceability weighted by vulnerability and area of feature remaining. This provides a better spread of irreplaceability values as opposed to other irreplaceability measures available in C-Plan that tend to clump values in specific value classes.

The patterns of irreplaceability in Figure 6.4 reflect the degree of transformation of features and number of features present in a planning unit. Areas of highest irreplaceability are all associated with the areas of cropping agriculture in the south west of the planning domain, the Bokkeveld escarpment and Sandveld. In these areas almost all remaining natural habitat is required to achieve the targets set. The next highest categories of irreplaceability highlights areas that have many features such as topographic refugia, riparian corridors or edaphic interfaces in addition to vegetation type and land-class features. As the targets for these process features were set relatively high, most planning units that have some of these features present are required to meet the targets set. White areas in Figure 6.4 have a summed irreplaceability of zero as these planning units are covered entirely by agricultural fields and therefore do not contribute anything to achieving targets.

Some sites are mostly transformed and contain very little natural habitat. To gain a better idea of the options for achieving targets majority transformed sites (>60% transformed) were removed from the set of available sites (Figure 6.5). Included in these excluded sites are transformed areas not indicated in the SKEP transformation map such as the Namakwa Sands mine site at Brand se Baai. In the design of the biosphere reserve these sites should ideally be excluded from any of the biodiversity compatible land-use zones. For a relatively untransformed landscape such as the Knersvlakte this is a sensible step to make before embarking with the reserve design, as there are still options for achieving targets elsewhere and there is no need to include these areas when selecting areas to achieve targets. This approach breaks down for landscapes or individual features that will require restoration of transformed areas if their targets are to be met. There are such examples in the south of the planning domain (Chapter 6.5).

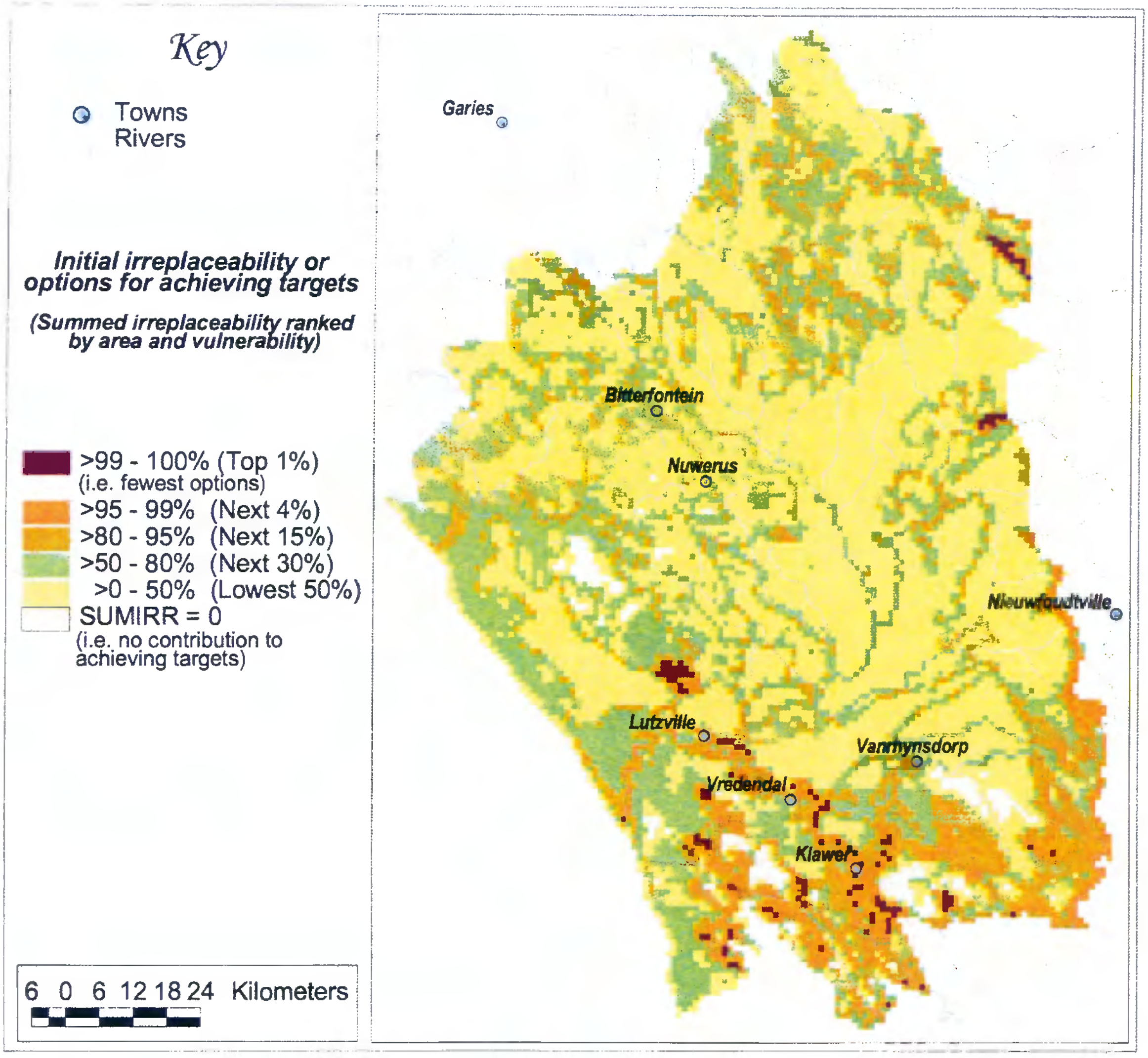


Figure 6.4: Options for achieving the biosphere reserve targets based on summed irreplaceability. The viewshed target is not included here. White areas (summed irreplaceability = 0) are transformed areas that do not contribute any natural habitat towards achieving targets.

By including the existing and proposed reserve network the map of options is further modified. When these are included in Figure 6.6 there does not appear to be a dramatic change in how the map looks. This is illustrates the point made previously that it is very difficult to achieve large-scale process targets within statutory reserves. If the targets were adjusted to representation targets (i.e. biodiversity pattern targets), which are much lower than the process targets (see previous Chapter for examples), then this map would look significantly different, as these reserves would achieve targets for some features. The core reserve does achieve representation targets for at least 11 features. This, however, is not the purpose for the biosphere reserve and no map showing representation target site irreplaceability is presented her. Figure 6.6 demonstrates and reiterates that an integrated landscape level approach is required to addressing the problem of the persistence of biodiversity.

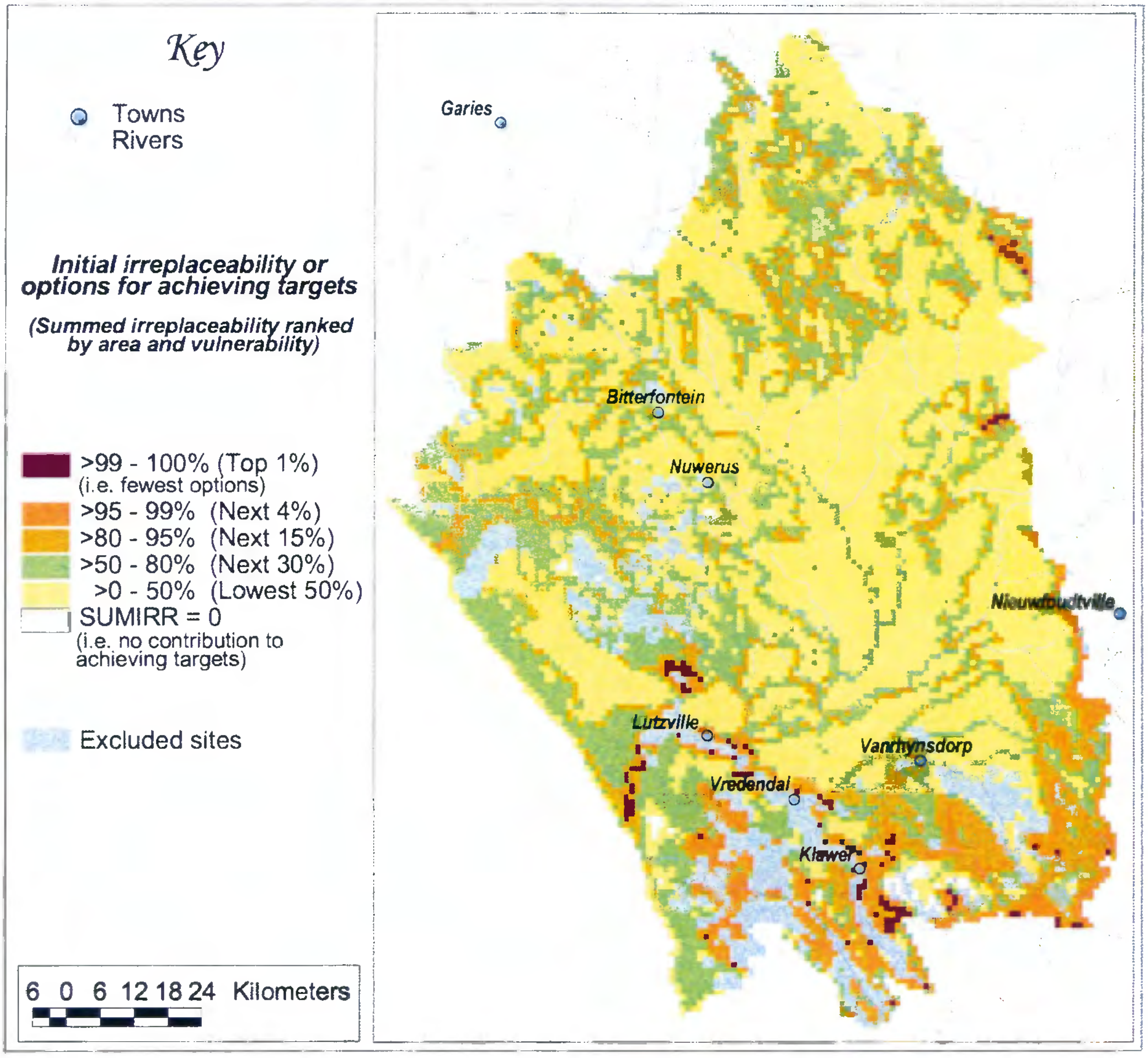


Figure 6.5: Options for achieving biosphere reserve targets when sites that have been more than 60% transformed have been removed from the pool of sites available for conservation. Here viewshed is included as a feature.

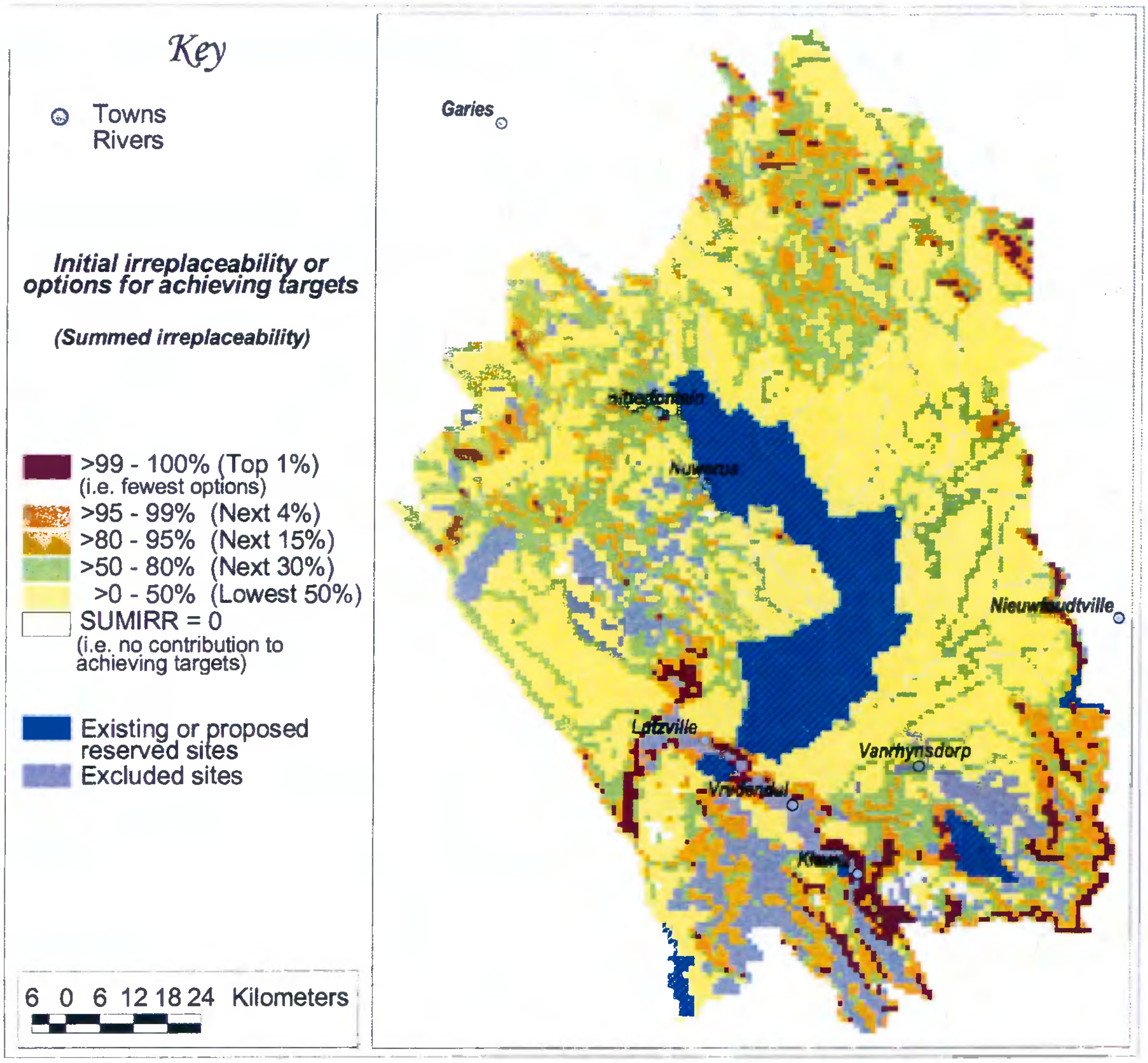


Figure 6.6: Options for achieving biosphere reserve targets when existing reserves and proposed reserves are included. As in the previous figure, the viewshed is excluded as a feature.

6.4.2 Biosphere Design Step 2: The minimum set.

Moving from the map of options to a reserve outline is not a simple process. In a planning domain with 13 556 sites and 156 (157 including the viewshed) features deciding what to do or where to go can be confusing. Where options are limited, i.e. high irreplaceability sites, these sites need to be included if targets are to be met, and there are few choices that need to be made. In transformed landscapes making conservation choices is made easier by the fact that there are few or no choices. At the other extreme where there are many options, i.e. at the bottom of the irreplaceability scale, there are numerous options as to which sites to select that complicate choices.

To facilitate this task of selecting areas a minimum set can be used to let the software decide what would be the most area-efficient means of achieving all targets. Figure 6.7 is an example of a minimum set that meets all targets (excluding viewshed). Normally, the outcome of a minimum set will vary with successive iterations using the same rules especially where there are many options for achieving targets. Using C-Plan, however, the same outcome is always achieved as C-Plan automatically selects the first site when there is a tie between sites rather than selecting a site randomly. There is no option in C-Plan to override this minimum set procedure rule. Where there are limited options to achieve targets, such as in the south, sites will always be included in the minimum set outcome.

The minimum set (Figure 6.7) illustrates what the most spatially efficient biosphere reserve could look like. The minimum set selects 61% of the landscape in addition to the 9% already "reserved" to achieve all targets (Table 6.4). Out of the 156 features considered, targets for 17 could not be met (Table 6.5). These features all occur in the south around the lower Olifants River valley, coastal Sandveld and inland mountain ranges. These areas will require restoration of transformed areas if their targets are to be met.

For comparative purposes Figure 6.8 shows the same minimum set with the site irreplaceability or additional areas required to achieve the viewshed target. As the Knersvlakte landscape is most fairly flat, preserving the viewshed will require a significant area of the planning domain in addition to that needed to achieve biodiversity process targets.

There are no rules in the minimum set that promote adjacency or connectivity which means that selected sites tend to be scattered rather than grouped into a pattern resembling Figure 6.2. Any connectivity in the landscape in Figure 6.7 as a result of the minimum set is fortuitous. It is also interesting to note that the minimum set does not choose a "buffer" around the core reserve. Many features are already represented within the core reserve and so the minimum set tends not to select sites adjacent to the core reserve that share the same features as represented in the reserve – a problem of autocorrelation. Without some type of adjacency rule in the minimum set the minimum set will generally tend to select sites away from existing reserve rather than selecting sites bordering these reserves.

The next step of the planning process addresses the two problems with the design highlighted thus far. Firstly, adjacency and connectivity in the landscape needs to be improved to reduce the moth eaten design as a result of the minimum set. Secondly, where possible areas need to be selected that meet the viewshed targets as well as the biodiversity targets.

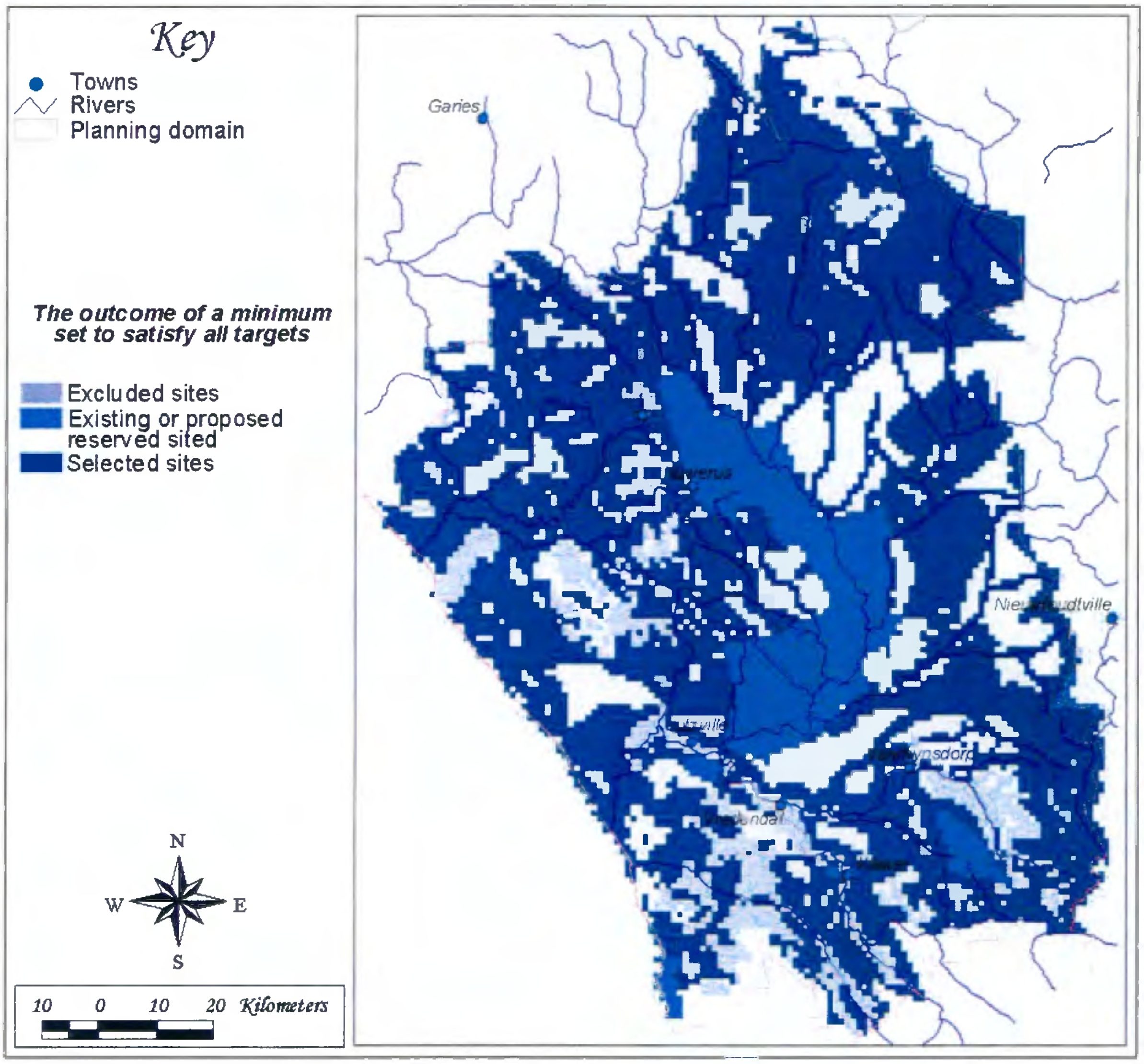


Figure 6.7: An example of a minimum set that satisfies targets for features excluding the viewshed. Sites were selected based on contribution to targets only and no adjacience or connectivity rules were used.

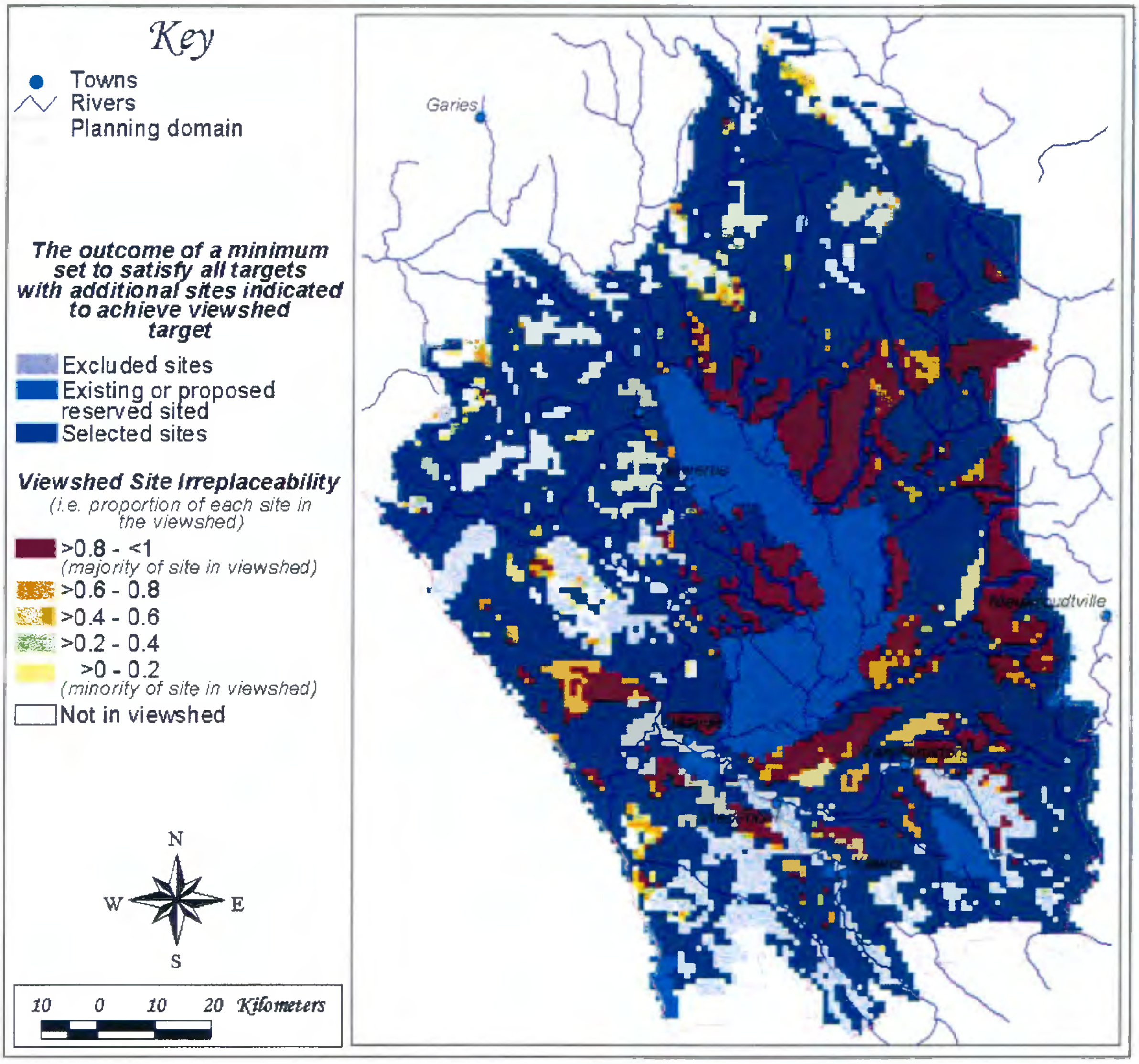


Figure 6.8: The same minimum set as in the previous figure with sites contributing to the viewshed target indicated. The flat Knersvlakte landscape means that much of the planning domain especially in the eastern half is visible form the core reserve.

6.4.3 Biosphere Design Step 3 to 5: Refine the reserve design.

The process of moving from the minimum set outcome to a more connected reserve involves the manual selection and de-selection sites to satisfy the design rules and viewshed targets whilst maintaining as best possible the percent target achieved by the minimum set for each feature. This is illustrated schematically in Figure 6.9. There are two potential trade-offs that result from this process. Firstly, to improve landscape connectivity may mean that more sites are required to achieve the feature targets set, i.e. the resultant design is less area efficient than the minimum set. Secondly, improving connectivity may mean that some targets cannot be met. This could arise in highly fragmented landscapes where remaining habitat is restricted to isolated patches that cannot be connected via corridors of natural habitat. This trade-off does not occur in the planning domain.

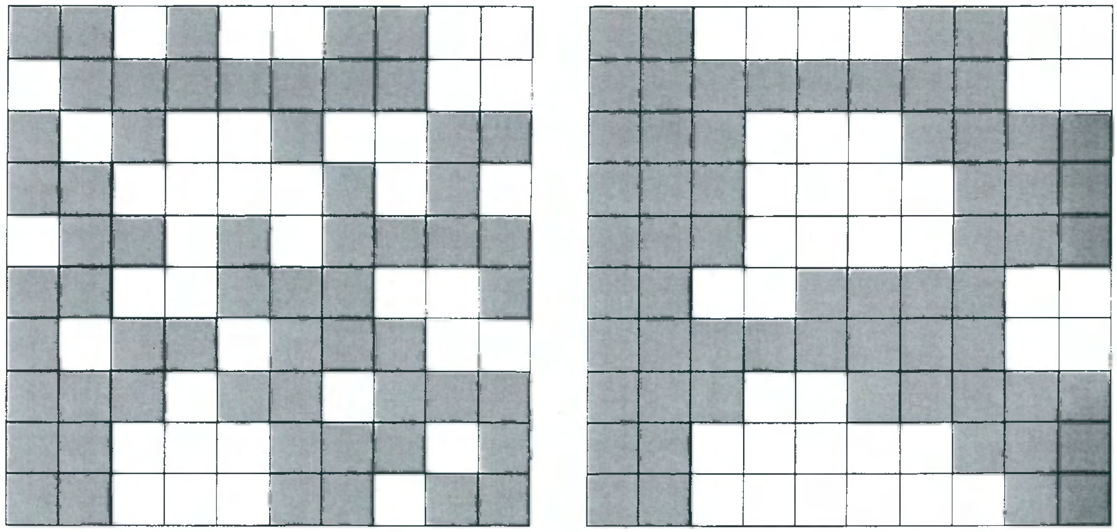


Figure 6.9 A conceptual illustration of the design process involved in moving from the minimum set outcome to a reserve design that has improved landscape connectivity. The illustration on the left is a potential minimum set outcome that selects 60% of sites to achive all feature targets. The illustration on the right shows the outcome of the manual design process that attempts to improve landscape connectivity. The trade-off of this process in this case is that more sites are required to achive the same percentage target achieved by the minimum set.

In the south of the planning domain, the viewshed was used to guide which sites would be selected in cases where the options for achieving targets were numerous. The rapid expansion of cropping agricultural in the southern part of the planning domain means that options for preserving the core reserve viewshed are retreating rapidly. The WODRIS agricultural potential map is a fine-scale map (1:10 000) that identifies those areas most sought after for cropping agriculture in the lower Olifants River valley area. This map was also used to help guide selections with high potential agricultural areas being avoided where possible whilst still achieving biodiversity targets, design rules and the viewshed. In the south existing agricultural development has already compromised conservation targets (Table 6.5). By accommodating agricultural development needs in the design process does not reduce any other feature's extent to below target in addition to the 17 already impacted.

There are options in C-Plan to incorporate competing land-uses into the feature-set so that the software can help make the biodiversity-alternative land-use trade-offs. In this project it was decided to manually incorporate this data at this stage as the area of conflict forms a relatively small part of the planning domain. For implementation of the biosphere reserve it may be a worthwhile exercise to perform another planning study specifically for the Lower Olifants River area at the 1:10 000 scale. In this area very small changes in boundaries can have significant conservation and economic trade-offs especially considering that agricultural land prices here are probably some of the most expensive in the Western Cape Province (Chapter 6.5.4).

Table 6.4: A comparison of the percentage area in each land-use category identified by the minimum-set versus the biosphere reserve.

Biosphere Reserve Categories	Planning Status	Minset	Biosphere
Biosphere Categories A, B and Ca		**71.01**	**70.53**
	Selected sites	*61.35*	*60.87*
	Initial or proposed reserves	*9.66*	*9.66*
Biosphere Categories Cb, D and E		**28.99**	**29.47**
	Initially excluded sites	*8.48*	*8.48*
	Unselected sites	*20.51*	*20.99*

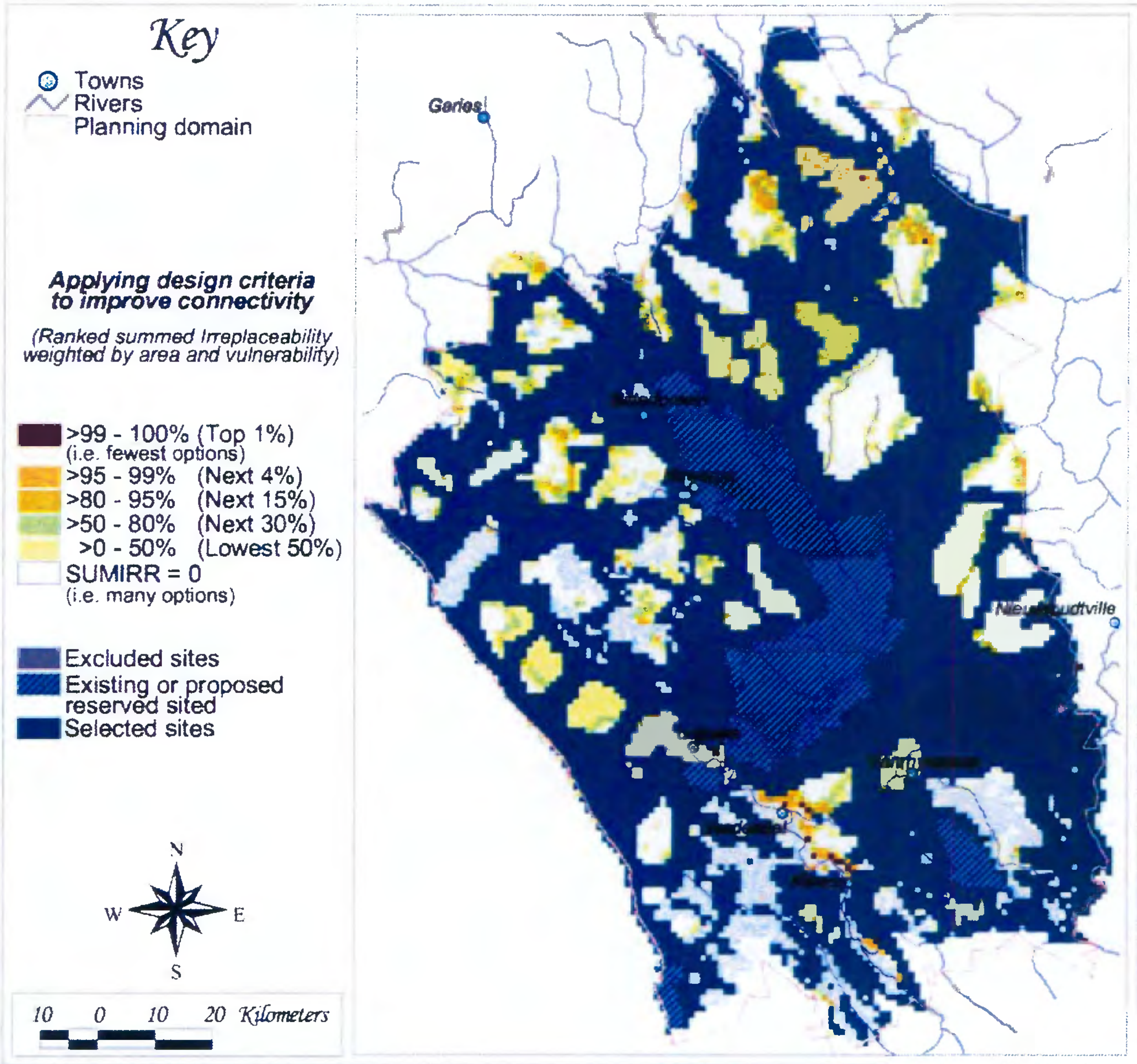


Figure 6.10: Applying design criteria to the minimum set to improve connectivity in the landscape.

Table 6.5: Percentage of each target met with the minimum set versus the biosphere reserve (including the core reserve). Features highlighted in blue are features whose target cannot be met as transformation has reduced the extent of these features to below that targeted. The viewshed highlighted in yellow was not included in the feature set used for the minimum set or biosphere reserve. The five features highlighted in green are features whose target was met by the minimum set but were not met by the biosphere reserve.

Name	Minset	Biosphere	Name	Minset	Biosphere
SW intermediate heuweltjie/quartz veld	39.74	39.74	Koekenaap Quartz Patches	133.01	133.01
River Order 5 (buffer 500m)	45.56	40.89	Land Class 40	134.26	151.74
SW quartz patches	70.84	60.07	Land Class 50	134.47	144.32
Viewshed	74.11	77.8	Knersvlakte Quartzfields	134.79	152.33
Olifants River Quartz Patches	75.13	44.06	Land Class 52	135.57	118.16
Leipoldtville Sand Fynbos	79.73	79.6	Land Class 29	136.2	134.02
Namaqualand Alluvia	88.66	76.57	Land Class 34	136.29	121.3
Plant pgd41	89.3	89.3	Land Class 54	136.33	132.28
River Order 1 (buffer 50m)	90.44	81.72	Land Class 31	137.48	115.18
Edapic interfaces (buffer 500m)	92.24	75.78	SE intermediate heuweltjie/quartz veld	138.33	130.3
River Order 3 (buffer 200m)	92.99	92.74	Troe-Troe River Quartz Patches	138.59	111.88
River Order 2 (buffer 100m)	93.26	91.96	Land Class 49	139.29	157.51
Fish f4	96.34	94.81	Land Class 4	140.07	142.55
Topographic climate refugia	97.15	87.98	Land Class 48	140.17	118.15
South-facing slopes climate refugia	97.81	88.77	Land Class 55	140.92	148.5
Graafwater Sandstone Fynbos	98.05	98.05	Land Class 33	141.56	128.9
Sea-facing topographic climate refugia	98.54	89.37	Plant ms3	141.82	193.87
River Order 4 (buffer 250m)	99.57	98.76	Rooiberg Quartzite Succulent Karoo	143.33	138.09
Agter-Sederberg Succulent Karoo	100	100	Birds b22	143.37	146.21
Plant ms6	100	100	Land Class 47	144.05	123.97
Invert JI32 Marble hills	100	100	Land Class 66	144.34	148.37
Invert JI33 Kommandokraal	100	100	Land Class 24	145.25	123.07
Namaqualand Red Sand Plains	100.02	101.76	Land Class 17	146.79	120.81
Northern Knersvlakte Lowland Succulent Ka	100.07	99.99	N quartz patches	146.94	149.57
Knersvlakte Shales	100.11	113.02	Central quartz patches	147.06	165.63
Bokkeveld Sand Fynbos	100.12	100.29	Land Class 27	152.51	129.1
Namaqualand Spinescent Grasslands	100.14	145.6	Central intermediate heuweltjie/quartz ve	153.17	165.22
Southern Knersvlakte Lowland Succulent Ka	100.15	100.58	Land Class 64	153.9	128.75
Lamberts Bay Strandveld	100.19	105.34	Limestone	154.94	156.91
Namaqualand Sand Fynbos	100.26	99.33	Land Class 79	155.03	128.22
Remhoogte Quartz Patches	100.5	155.24	Land Class 2	155.74	148.63
Namaqualand Southern Strandveld	100.6	131.79	Land Class 19	155.93	141.23
Plant ms5	101.07	128.17	Land Class 67	156.4	103.55
Namaqualand Lowland Succulent Karoo	101.8	84.67	Vanrhynsdorp Shale Renosterveld	156.57	155.75
W intermediate heuweltjie/quartz veld	102.2	166.67	Land Class 12	157.54	145.51
W quartz patches	102.27	166.67	Land Class 74	157.54	166.67
Doring River Succulent Karoo	103.41	104.99	Land Class 6	157.9	139.62
Land Class 35	105.09	130.79	Land Class 91	158.05	120.69
Land Class 3	106.28	110.69	Knersvlakte Dolorites	159.03	109.63
Plant ms4	106.47	122.84	Land Class 43	159.84	162.74
Plant pgd18	107.79	132.6	Nuwerus Quartzite Succulent Karoo	160.34	148.08

Name	Minset	Biosphere	Name	Minset	Biosphere
Land Class 21	107.89	105.76	Land Class 73	161.25	160
Land Class 7	109.85	125.46	Land Class 9	161.5	147.49
Platbakkies Quartz and Gravel Patches	110.07	110.07	Kamiesberg Mountain Brokenveld	162.06	149.75
Land Class 38	110.91	130.2	Land Class 86	163.62	163.58
Land Class 76	111.29	118.49	Land Class 42	163.9	154.38
Central Knersvlakte Lowland Succulent Kar	111.4	136.08	Land Class 18	164.15	151.75
SE quartz patches	112.01	107.11	Land Class 16	164.72	160.76
Land Class 63	112.48	143.49	Land Class 59	165.18	163.43
Namaqualand Klipkoppe	114.76	100.19	Land Class 78	165.34	143.35
Land Class 46	114.89	119.12	Land Class 10	165.66	130.43
Land Class 82	115.99	108.33	Hantam Karoo	166.67	116.24
Komkans Quartz Patches	116.17	106.76	Kotzerus Quartz Patches	166.67	86.42
Namaqualand Arid Grasslands	116.79	107.71	Land Class 5	166.67	165.01
Land Class 8	117.57	106.13	Land Class 28	166.67	156.77
Land Class 30	118.34	114.8	Land Class 41	166.67	166.67
Land Class 45	120.13	166.67	Land Class 44	166.67	154.48
Namaqualand Klipkoppe Flats	122.82	97.82	Land Class 65	166.67	156.73
Arid Coastal Salt Marshes	122.82	122.93	Land Class 70	166.67	132.18
Bushmanland Basin	123.87	100.35	Land Class 85	166.67	164.08
Land Class 61	124.7	147.78	Plant pgd17	168.27	190.98
Land Class 77	124.84	112.02	Fish f2	190.49	169.82
Land Class 56	125.82	153.12	Plant pgd16	196.55	200
Land Class 13	126.53	139.06	Fish f3	198.92	200
Land Class 22	126.69	124.93	Plant pgd20	476.32	733.41
Land Class 39	128.14	123.22	Plant pgd19	556.8	547.18
N intermediate heuweltjie/quartz veld	128.82	133.42	Plant pgd22	609.97	833.33
Land Class 80	130.71	128.9	Plant pgd14	744.03	881.65
Land Class 32	130.97	119.05	Amphibian ao7	745.21	789.03
Land Class 26	132.2	101.97	Amphibian ao6	790.42	710.1
Quart-patch areas	132.47	141.75	Plant pgd15	799.25	816.29

6.4.4 Biosphere Design Step 6: Relate design to actual boundaries.

It is difficult to relate the square planning units used to develop the biosphere reserve to where actual boundaries might be located on the ground. In many cases planning units are partly transformed and generally these areas would be excluded from the conservation zones of the biosphere reserve. To ease interpretation of the biosphere boundaries the planning units were converted to polygons whose boundaries relate to underlying features, as they are located in the landscape. This process involved overlaying the map of selected sites on the satellite image and manually digitizing the boundaries in relation to the observed vegetation patterns, landscapes features and transformation (Figure 6.11).

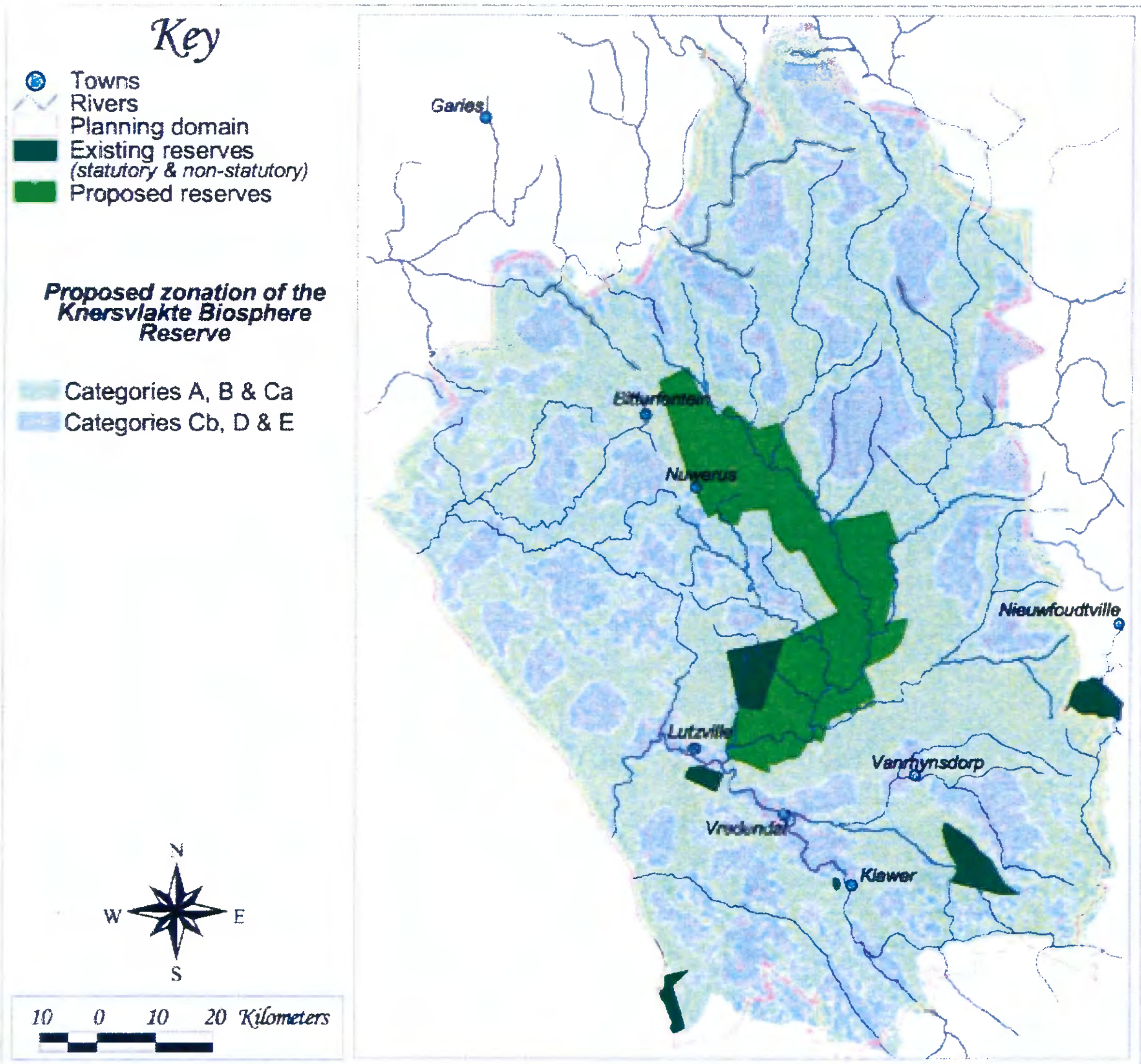


Figure 6.11: Broad categorisation of landuse in the Knersvlakte Biosphere Reserve.

The product of the previous stage must also be interpreted in terms of proposed land-use planning categories. The existing and proposed reserves and the selected sites together constitute the biosphere reserve categories A, B and Ca. The initially excluded sites and sites not selected together constitute the biosphere reserve categories Cb, D and E. This is a preliminary broad categorization of the landscape and no further subdivision in terms of these categories is attempted at this stage. It must be remembered that apart from the core reserve identified in this study and other existing reserves, no additional core conservation areas are identified here. Identification of these will need to form the focus of another study. Also, the question of restoration of vegetation types or land-classes that have been transformed to below their process targets is not addressed here. This is discussed further in the following section.

6.5 Setting Priorities of the Biosphere Reserve

All areas in the landscape are important for the preservation of biodiversity whether directly through the species present or indirectly through the processes supported. The biosphere reserve design is not a map of important areas for conservation. It is a map of what is estimated as the minimum requirement for the persistence of biodiversity in the landscape. Thus, in the context of the biosphere reserve, all areas mapped as categories A, B or Ca are equally important because they all contribute to achieving conservation targets. Even areas that may appear to be biologically boring are important within the context of the biosphere plan, as they contribute to achieving the conservation targets.

The irreplaceability maps show that for many areas in the biosphere reserve there are options as to where targets are achieved. These maps also show that in other areas there are few, if any, options available to achieve targets. The whole biosphere reserve will not be implemented at once. Thus, priorities are necessary, firstly, to act in those areas that are most important from a biodiversity perspective (e.g. conserving populations of threatened species), but more importantly to act in areas where options for achieving goals are limited or retreating due to the impacts of alternative land-uses.

Three methods are employed here to help assess the priorities for implementation of the biosphere reserve. These methods are:

1. Prioritize sites based on the irreplaceability of the site versus the potential for that site to be converted to alternative land-uses (i.e. irreplaceability vs vulnerability).
2. Prioritize biodiversity features based on the pattern target set for those features and the degree to which they have been transformed (i.e. conservation status).
3. Use expert knowledge to integrate information not captured in the datasets to decide on priorities.

The priority analysis presented here should be regarded only as a general guide to priorities for the implementation of the biosphere reserve and should not be regarded as prescriptive. When the planning outputs of this project eventually get to the implementation phase of the biosphere reserve these priorities should be reassessed in conjunction with stakeholders in order to accommodate limitations and opportunities presented by the different organizations involved in the process.

6.5.1 Irreplaceability vs vulnerability

The basic irreplaceability-vulnerability analysis in conservation planning is a recognized technique for assessing priorities (Margules and Pressey 2000b; Pressey and Taffs 2001b). Depending on where sites fall on the two axes determines how they will be approached during the implementation phase of the conservation plan. Basically, sites with highest irreplaceability and the highest vulnerability to being transformed constitute priority areas for conservation action. These are sites where there are few options for achieving targets, i.e. that site needs to be conserved in some form or another now if the conservation targets are to be met. If there is no action in the short term there is a high potential that that site will be lost to some biodiversity incompatible land-use.

Table 6.6: Three action categories for the biosphere reserve implementation based on the combination of options for achieving targets (irreplaceability) and alternative land-use potential (vulnerability). Area covered by each land-use rank was summarised for each site, and the site assigned the highest rank of the two land-uses that covered ten or more percent of the site.

	Agricultural Potential			Mining Potential			
Option category *(Irreplaceability)*	Low	Medium	High	Low	Medium	High	Very High
Top 25% of sites	**LOW**						
Next 25% of sites		**MEDIUM**					
Last 50% of sites							

Sites located elsewhere on the two axes have different associated implementation requirements. For example, a site with high irreplaceability but low vulnerability is required to achieve targets but action is only necessary in the medium to long term as there are no other land-uses competing for that same land. Likewise, a site with low irreplaceability but high vulnerability means there are several sites that will achieve the same target and if this site is lost in the short term it is not a crisis as there will be alternative sites. How implementation deals with different combinations of irreplaceability and vulnerability in terms of scheduling of action or type of action undertaken needs to be addressed in a context specific manner and is not discussed further here.

With over 13 000 sites, plotting irreplaceability versus vulnerability for the biosphere reserve would not provide meaningful insight. Instead the possible combinations of irreplaceability and vulnerability were summarized into three potential action classes and then plotted on a map to show the priorities spatially.

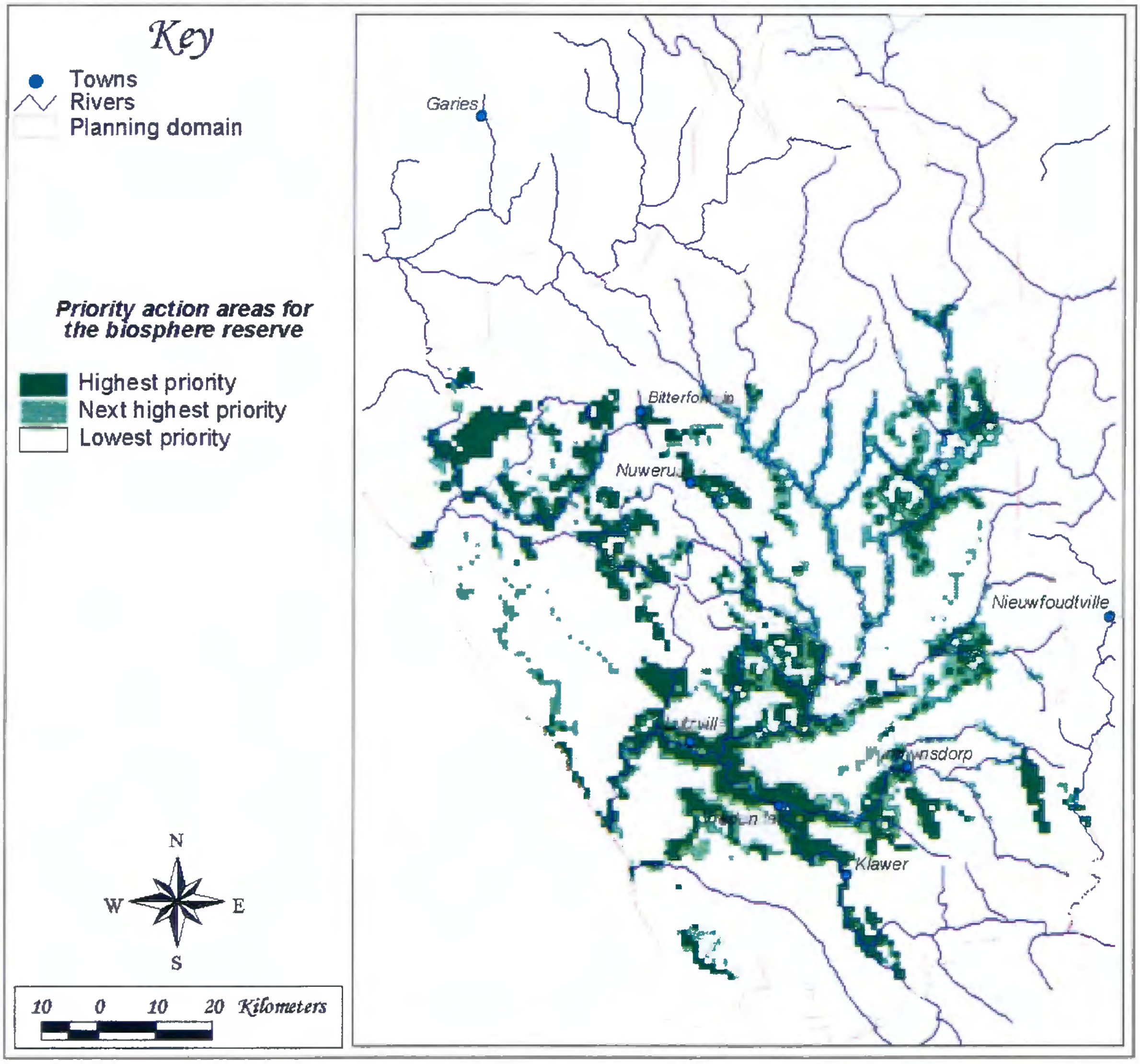


Figure 6.12: Priorities for action in the biosphere reserve based on available options to achieve conservation targets (irreplaceability) and potential to support alternative land-uses (vulnerability). See Table 6.6 for how the action categories relate to irreplaceability and vulnerability.

In Figure 6.12 priorities for implementation of the biosphere reserve are located mainly in the south and along the northwest hardeveld klipkoppe. In the Sandveld south of Vredendal; the lower Olifants River Valley; and, the Olifants River, Gifberg and Bokkeveld mountains areas targeted by agriculture are prioritized. North of the Olifants River priorities are in areas targeted by mining, e.g. granite koppies or limestone.

In the south options for achieving targets are most constrained by existing transformation. Future development in this area needs to be within a framework of sustainable developments otherwise the persistence of the regional biodiversity will be compromised. In the northwest, priorities are driven by the high targets set for certain mountainous habitats associated with buffering landscapes against climate change.

When interpreting Figure 6.12 it is important to be aware of the limitations of the data used. The minerals potential data does not extend much further north than Bitterfontein. Also the agricultural potential data does not extend south of approximately Strandfontein for lowland areas and does not include the upper Olifants River, Gifberg or Bokkeveld mountains. The trends in priorities in Figure 6.12, i.e. associated with transformed areas in the south and mountainous habitats in the north, will probably be extended to those areas where there was no alternative land-use data.

6.5.2 Conservation status

Another method for identifying priority areas for action is to rank vegetation types based on their degree of transformation. This means that priorities for conservation action are based on the degree to which features are transformed in relation to the targets set for these features. Transformation of a biodiversity surrogate such as a vegetation type correlates with loss of biodiversity. Features that are highly transformed are those where there are fewest opportunities to achieve conservation targets. Failing to act promptly in these areas might mean that conservation targets may never be achieved, if not already. At the other end of the transformation spectrum where a feature has been little transformed there are many options for achieving targets therefore where resources for conservation action are limited or where action is aimed at minimizing further loss of biodiversity these features are not necessarily priorities for action.

Overall transformation of natural habitat within the planning domain is low with only approximately 11.8% of the area being transformed. This transformation, however, is not evenly spread between features (**Error! Reference source not found.**Figure 6.13). There are nine features that are more than 40% transformed and only two that are more than 60% (Figure 6.14), the SW intermediate heuweltjie/quartz veld (a focus habitat type) located in the Lower Olifants River Valley and Land-class 73 located in the Gifberg. Other features are approaching this 40% transformed threshold, namely the Agter-Sederberg Succulent Karoo, Namaqualand Alluvia and Leipoldville Sand Fynbos vegetation types; the SW quartz patches focus habitat type; and, land-class 86. All these features are located within the major cropping agricultural areas of the planning domain. As can be expected, transformation is greatest in areas with higher agricultural potential such as the Bokkeveld Mountains, Olifants River valley and Sandveld south of Vredendal.

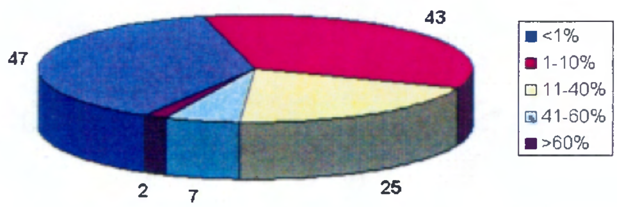


Figure 6.13: The percentage area of each feature transformed summarised for the 124 biodiversity pattern features (focus habitats, vegetation types and land-classes) in the planning domain. The numbers opposite each segment are the number of features in each transformation category

Using the pattern and process targets that were set in this study, biodiversity features (vegetation types, land-classes and focus habitat types) can be categorised into four different "conservation status" classes based on their respective levels of transformation (Figure 6.15). These categories can equate to the urgency for conservation action to prevent further loss of biodiversity. Note that the conservation status concept used here is based on numerous discussions conservation planning researchers and practitioners from the Institute for Plant Conservation, the Botanical Society, National Botanical Institute, Terrestrial Ecology Research Unit and Wolfe and Associates.

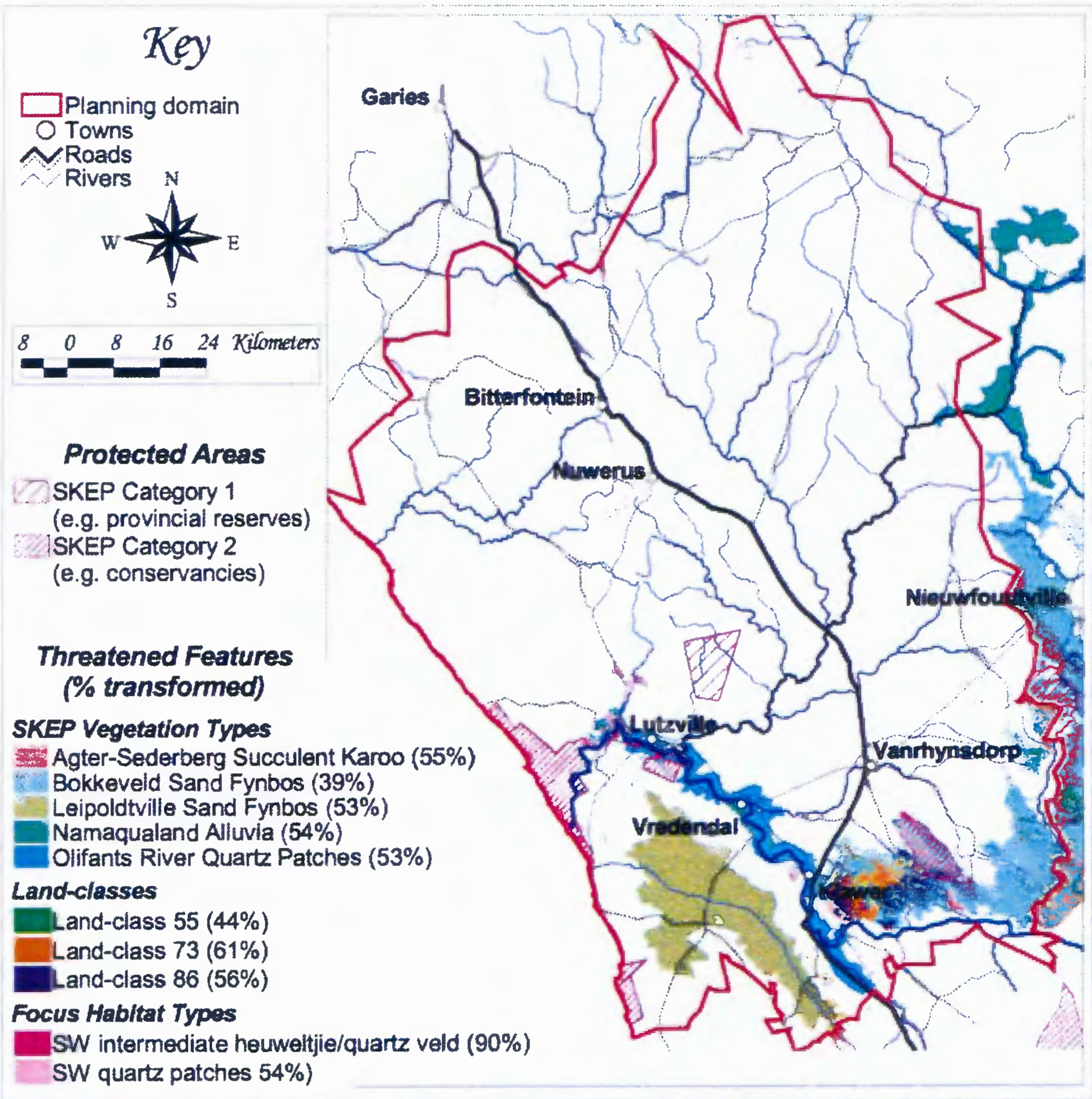


Figure 6.14: Biodiversity features (vegetation types, land-classes, and focus habitat types) that are threatened with transformation in relation to existing and proposed conservation areas in the planning domain. The location of threatened features is strongly related to the location of areas of high agricultural potential.

Firstly, biodiversity features that are transformed to the level of their pattern target (i.e. between 60% and 85% transformed depending on feature) are critically endangered (Figure 6.15). In these features natural habitat has been reduced to such an extent that they are below a critical area required simply to represent the majority of species. These features have already lost biodiversity and are likely to loose more in the short to medium term even if no further transformation occurs as they are below the important ecological process threshold. They have also lost to some degree their capacity to provide useful ecological services. This leads to the second category.

Biodiversity features that are more than 40% transformed (i.e. less than 60% of original extent remains, but more than their pattern target remains) are below the ecological process target set in this study. Biodiversity in the remaining natural habitat is at risk, as ecological processes are not functioning properly. There is a real chance that biodiversity will be permanently lost from these areas in the short to medium term. These biodiversity features are classified as endangered (Figure 6.15).

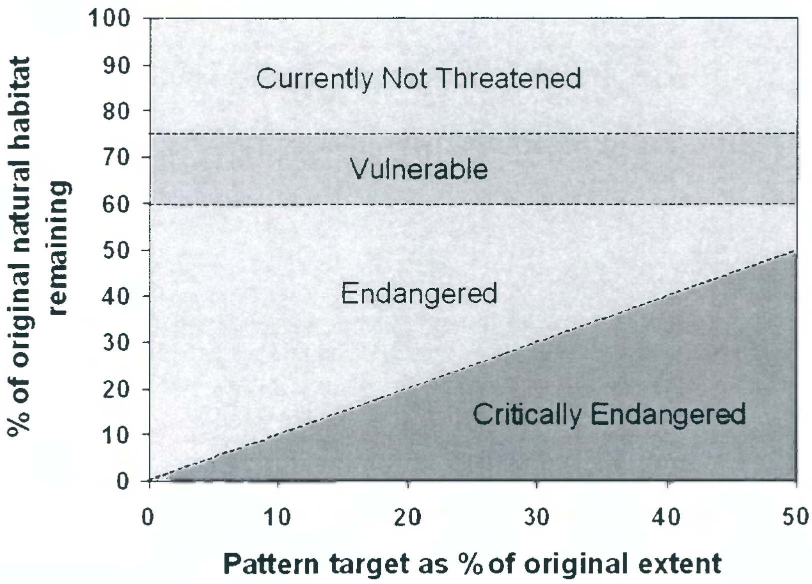


Figure 6.15 The definition of conservation status categories based on the relationship between pattern targets set for features and their degree of transformation. Figure adapted from AT Lombard unpublished data.

The third conservation status category applied to features that are approaching the 60% ecological process target. These features are currently above this threshold, however, future transformation could push these features over this threshold. It would be prudent to target these areas for conservation action in order to avoid these areas from being transformed beyond this threshold. These features are classified as vulnerable (Figure 6.15). The lower threshold for this category is 40% of a feature transformed, however, in this study there is no biologically based upper threshold for this category. For the present this threshold is set to 25% transformed making the "buffer" area of vulnerability 15%.

Developing a biologically sound rationale for the upper boundary of this conservation status category requires more thought.

The conservation status map for biodiversity features (vegetation types, land-classes and focus habitat types) shows, not surprisingly, that currently the most threatened biodiversity located in the south of the planning domain in areas of high agricultural potential (Figure 6.16 and Figure 6.17). These areas should form part of the initial focus of any effort to implement the biosphere reserve in the region. It must be noted, however, that these priorities are based on historical patterns of landscape transformation and not future potential for transformation. Thus, conservation status alone should not be used as the sole mechanism for prioritising conservation action. For example, Figure 6.17 does not indicate priorities in the core reserve where it is known that many areas are earmarked for future mining activities. In Section 6.5.1 setting priorities that incorporates vulnerability based on future potential for transformation is still a very useful additional mechanism for prioritising conservation action.

It should also be noted in Figure 6.17 that conservation status of features is based on their degree of transformation only within the planning domain. As the transformation map does not extend beyond the boundaries of the planning domain in this study it is difficult to say what the conservation status is of features that are highlighted outside of the planning domain. The results of the conservation status analysis should only be interpreted with respect to what is happening within the planning domain.

A fifth conservation status category has been proposed called a "protected ecosystem". This is any ecosystem that is of such high conservation value or national importance that it requires national protection, but which does not meet the criteria for threatened (critically endangered, endangered or vulnerable) ecosystems. Included in this category may be indigenous forests or quartz-patches. This category may be extended to include ecosystems that deliver important services for humans such as wetlands and riparian systems. There are no quantitative criteria yet as to what constitutes a protected ecosystem. Perhaps the best approach would be to used expert opinion as to what these may be. Although this category is not applied to the Knersvlakte study area, the following section could provide an initial basis for delimiting such ecosystems.

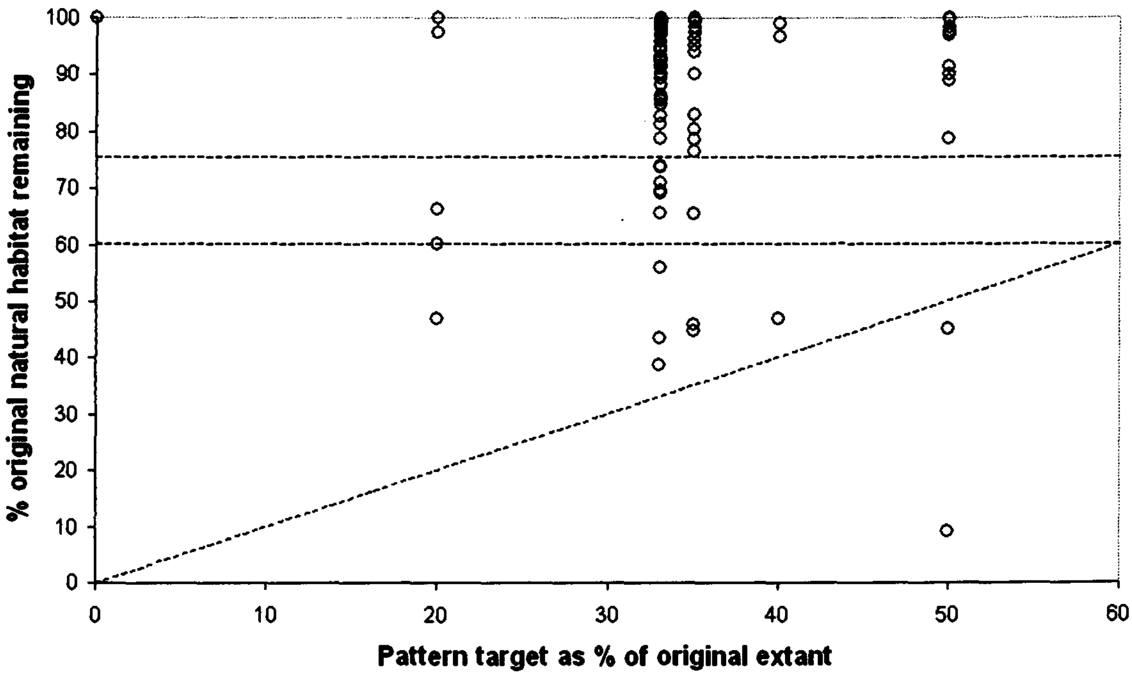


Figure 6.16 The conservation status vegetation types, land-classes and focus habitat types in the planning domain as defined in Figure 6.15.

6.5.3 Expert assessment

The third approach to identifying priorities is to use expert opinion to decide where key areas for action should be located. The rationale for using expert information is that it complements the existing data. Experts are often aware of recent developments or changes in the region that may affect outcomes that are not reflected in the quantitative analysis of the available data sets.

The expert assessment here is divided into three categories of based on the type of biodiversity feature involved and the nature or urgency of action required to safeguard the feature. These categories are:

1. Focus habitat types that are sensitive to land-use impacts;
2. Key corridors that are vulnerable to being lost;
3. Key nodes that are vulnerable to being lost.

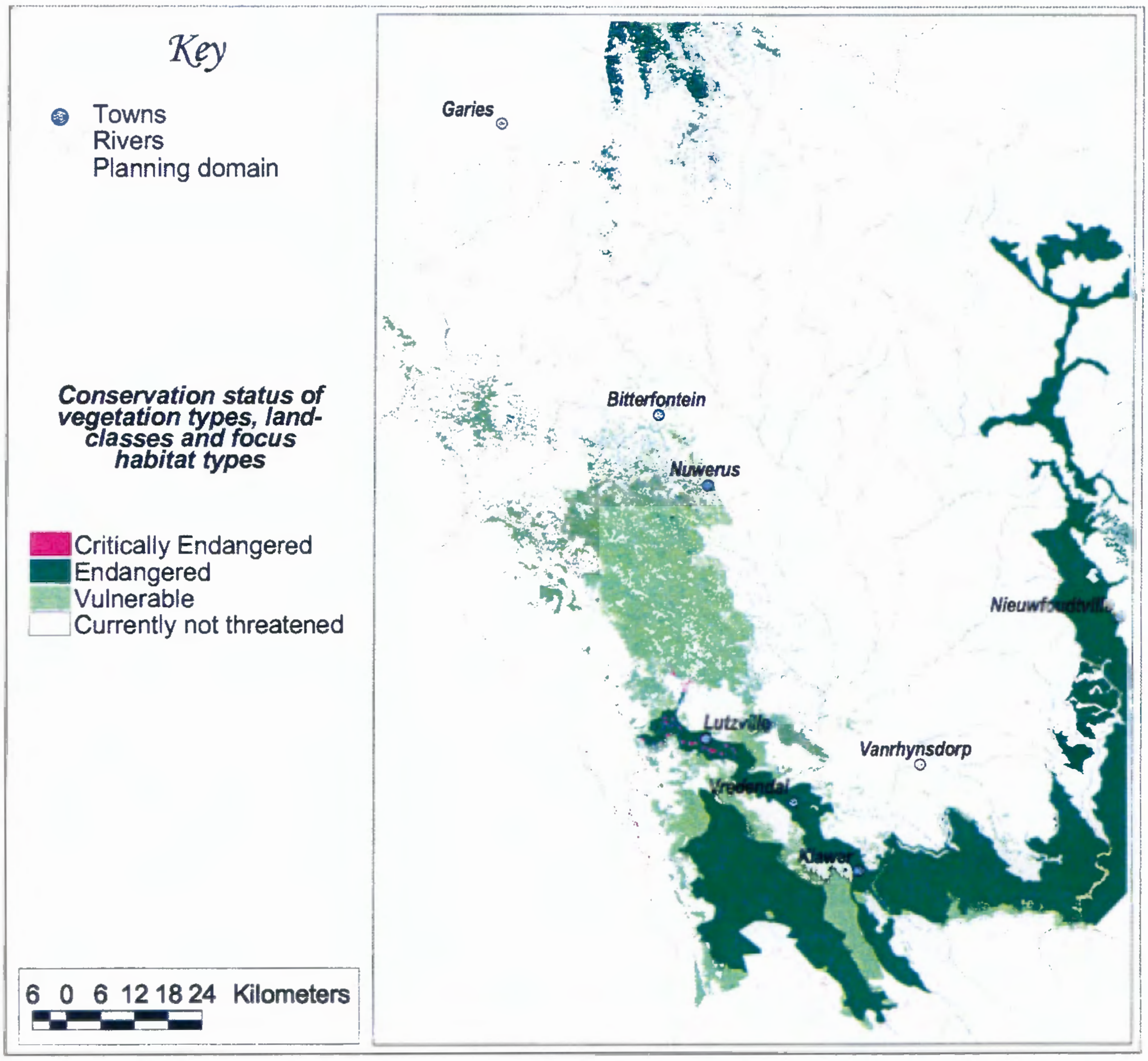


Figure 6.17: Conservation status of vegetation types, land-classes and focus habitat types based on patterns of transformation Red Data Book classification of vegetation types and sensitive areas in the planning domain.

6.5.3.1 Focus Habitat Types

The focus habitat types are the quartz-patches and limestone habitats whose biota is unique to the Knersvlakte. Quartz-patches are a feature of the Succulent Karoo biome and are a globally unique ecosystem. These habitats are sensitive to any high intensity land-use activity (Schmiedel 2002). These can have persistent negative impacts on the ecosystem. Also, mining in the area has shown that once these habitats are removed by mining they are lost forever. Anywhere where these habitats occur in the planning domain is classified here as a sensitive area because of these properties. This classification is based solely on ecological properties of the habitats and not vulnerability to transformation, as this is variable across the planning domain.

In Figure 6.18 the extent of these habitat in the planning domain is mapped. A 100m buffer is added to each patch of habitat to account for mapping errors and neighbourhood impacts of development that occurs near these habitats, i.e. the sensitive area is the patch of focus habitat type plus a buffer in the adjoining habitat type.

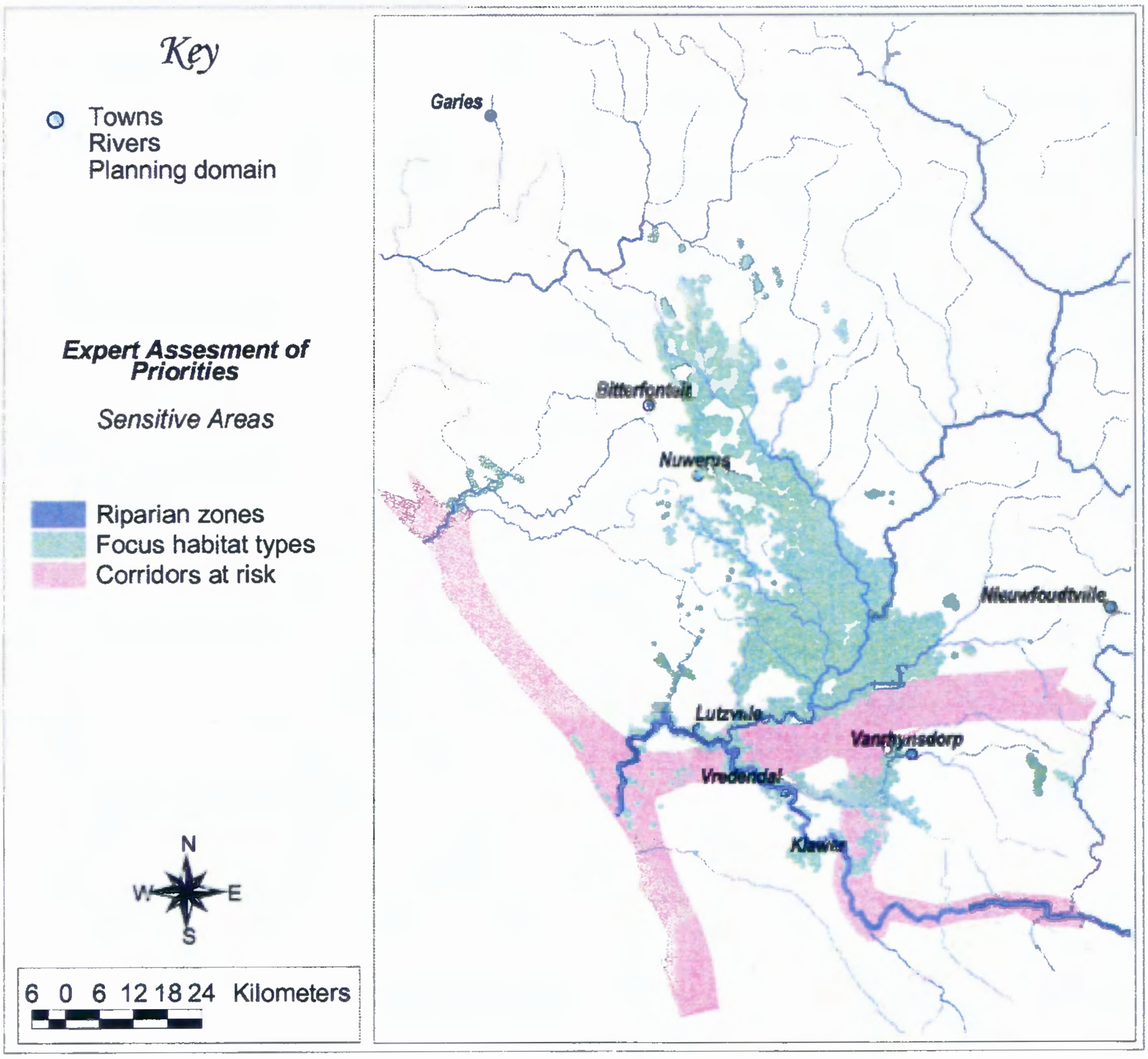


Figure 6.18 An expert assessment of priorities for the biosphere reserve based on the location of sensitive areas. These areas include the focus quartz-patch and limestone habitat types, riparian zones and ecological corridors at risk of being transformed due to alternative land-uses.

6.5.3.2 Key Corridors

Corridors are areas of landscape that connects two or more other areas in the landscape. They are generally seen as conduits for biota movement. A key attribute of any corridor is that it needs to provide biodiversity with a relatively uninterrupted passage across the landscape. In the planning domain any corridor that is vulnerable to being truncated is in danger of loosing its ecological function as a corridor and these are classified as sensitive areas. It must be remembered that corridors or linkages in the biosphere reserve plan connect or link all parts of the landscape. Only those that have been assessed as being at risk are mapped.

There are three such corridors identified in the planning domain (Figure 6.18). These are:

1. The entire coastal forelands strip is under pressure from mining and recreational activities. Of major concern are the mining activities at Namakwa Sands; and, the agricultural, recreational and mining activities in the area of the Olifants River mouth between Weskus Mynbou and Strandfontein. Elsewhere, but especially in these two areas provisions need to be made to ensure the maintenance of north-south biodiversity corridors.
2. The east-west spiny grassland corridor running for south of Ebenhaeser inland to east of the N7 is under pressure from cropping agriculture. Not only is this area south of the Varsch River a major component of the core reserve viewshed, but the deep sandy soils make it very attractive for irrigation agriculture. This is an important coastal inland Sandveld link that links the Strandveld at the coast with Bushmanland Sandveld grassland communities that enter the planning domain via the Sout River valley.
3. Riparian zones throughout the planning domain fall into this category of key corridors

6.5.3.3 Key Linkages

Key linkages are parts of corridors, but of all the features discussed in this section are the most vulnerable to being irreversibly lost in the short term. In the planning domain these are all located along the lower Olifants River valley and are connections that link the northern and southern flanks for the river. There are only four such linkages remaining across the Lower Olifants River (Figure 6.19). These are:

1. The river and estuary below Ebenhaeser
2. Where the Sishen-Saldanha railway crosses at Liebendal
3. Between Bruinkraans and where the N7 crosses the River, south of Klawer
4. Between Melkboom and the Bulshoek Barrage.

These are the only places where natural habitats on either side of the river remain relatively intact to the rivers edge. Elsewhere irrigation agriculture creates a relatively wide barrier to biotic movement. Loss of these connections will compromise the conservation vision for the landscape. These four areas should be the first implementation priorities for the Knersvlakte Biosphere Reserve.

6.5.4 Understanding Irreplaceability for Setting Priorities

Interpreting the outputs of this plan requires a better understanding of the determinants of irreplaceability. In different kinds of landscape irreplaceability needs to be interpreted differently for setting priorities for action.

Irreplaceability is a measure of conservation options. The higher a site's irreplaceability the more important that sites is in terms of achieving targets. It is intuitive to think that sites with higher "conservation importance" will have higher irreplaceability. This is not always so, and is demonstrated by site irreplaceability of the biosphere reserve. In Figure 6.4 it is surprising to see that areas of quartz-patches along the Sout and Geelbeks Rivers or Sandplain Fynbos along the coast have relatively low irreplaceability relative to areas, for example, along the lower Olifants River. Transformation of the landscape modifies how irreplaceability is interpreted in a planning context.

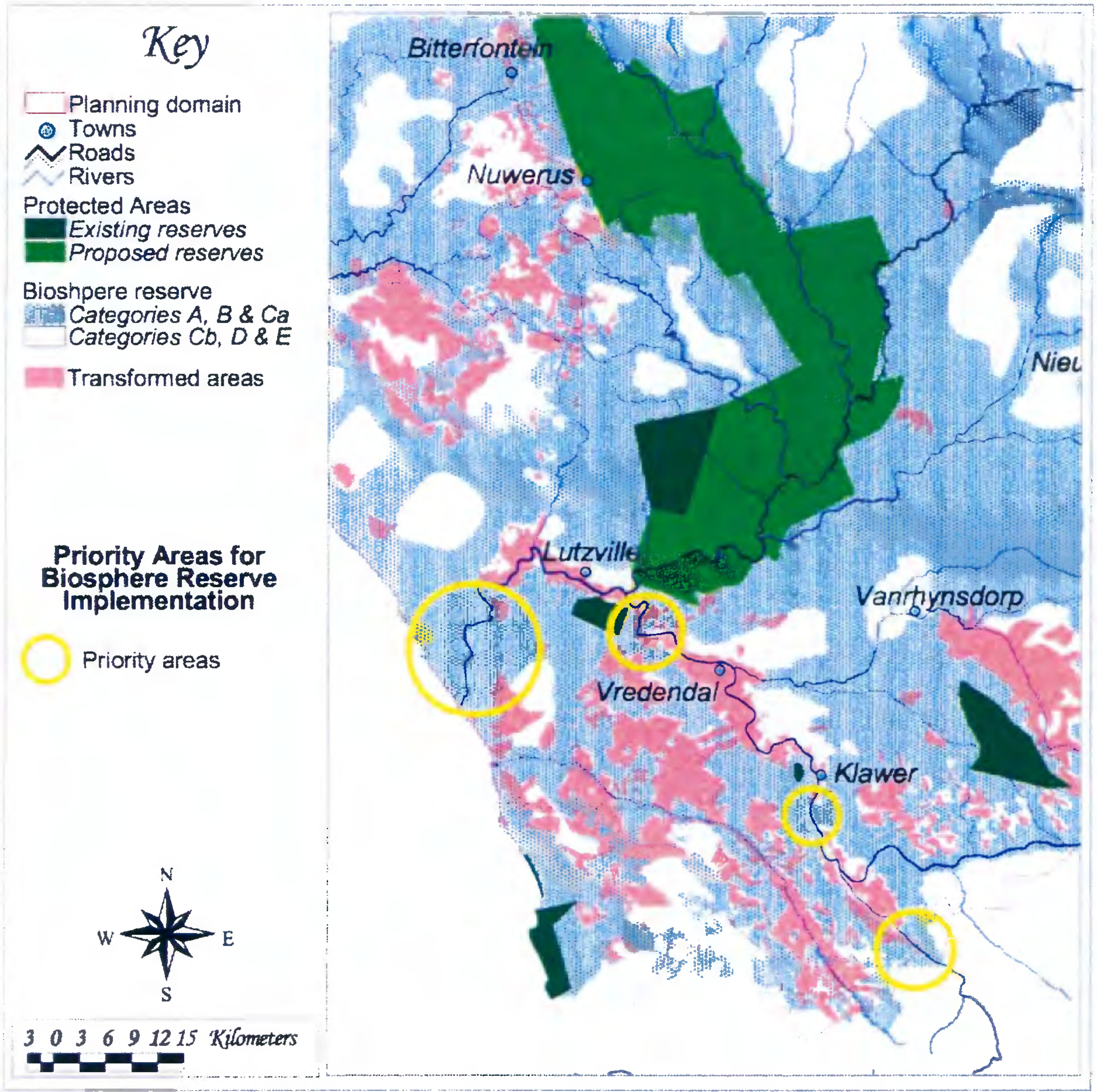


Figure 6.19: Expert assessment of priorities for biosphere reserve implementation highlighting the last remaining linkages across the Olifants River. All these linkages are highly vulnerable to being lost to cropping agriculture in the short term. Loss of these linkages will severely compromise the wider conservation vision for the landscape. These four areas should be the first priorities for implementation of the Knersvlakte Biosphere Reserve.

Comparing the relationship between site irreplaceability values and the number of features recorded per site, i.e. a crude measure of conservation importance (Figure 6.20a), between an area with high levels of landscape transformation, the lower Olifants River valley (Figure 6.22), and one with low levels of transformation, the southern

Kamiesberg mountains (Figure 6.22), shows that there is, as to be expected, a positive relationship between the two variables. However, there are on average five fewer features per site for any given irreplaceability value in the highly transformed landscape relative to the untransformed landscape. If number of features per site is a crude measure of "conservation importance" then interpreting irreplaceability as such will be misleading. Irreplaceability should never be interpreted as being a measure of "conservation importance" or "conservation value" (Pressey and Taffs 2001b).

There is generally a negative relationship between site irreplaceability and neighbourhood transformation (Figure 6.20b) in a low transformation landscape. This is perhaps what would be expected as transformed sites contribute less area of features to targets and would have lower irreplaceability. In a highly transformed landscape, however, the initial relationship between transformation and irreplaceability is positive. Here transformation has reduced some features to the extent that any site with some of those features present will be required to meet targets. Only once neighbourhood transformation exceeds 50% does the relationship begin to follow that of the low transformation landscape.

In landscapes characterized by high levels of transformation, high irreplaceability values will tend to be associated with moderately transformed areas. These are areas where the features that are being targeted by the agents of transformation still remain to some degree. In terms of conservation planning, priorities for action will invariably be associated with transformation. Taking this relationship further (Figure 6.21) irreplaceability correlates with land-value. Land that has transformation potential (i.e. mining and cropping agriculture) generally has a much higher value than land that is only suitable for stock farming (Figure 6.22). This is to be expected since irreplaceability is driven upwards as more land is transformed resulting in decreasing options for achieving targets. This trend also seems to hold for landscapes with low levels of transformation (Figure 6.21).

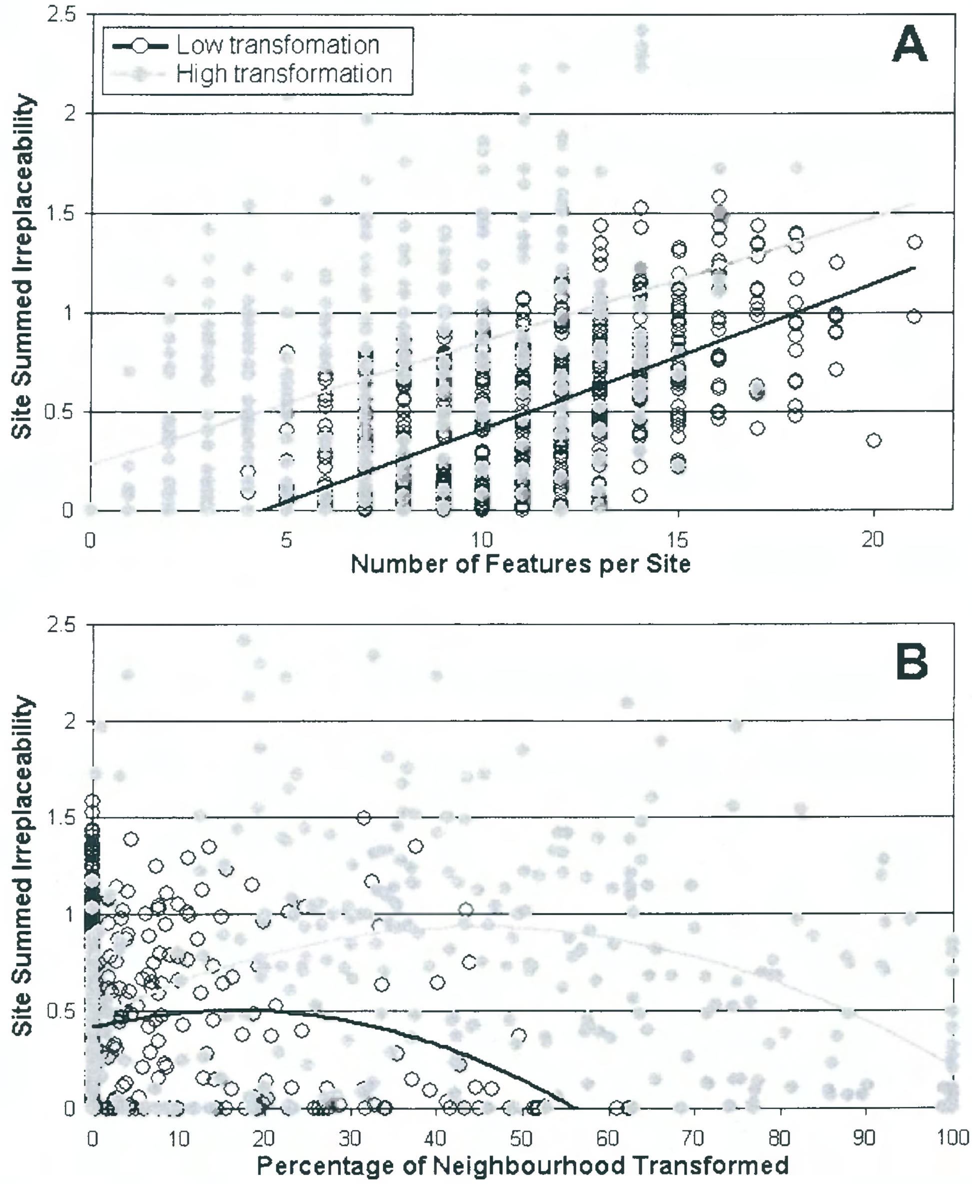


Figure 6.20: The relationship between conservation options (Site Summed Irreplaceability) and (a) the number of features per site and (b) the percentage of site neighbourhood transformed in a landscape characterised by low (A: $y = 0.0736x - 0.3211$; $R^2 = 0.4892$; B: $y = -0.0003x^2 + 0.0106x + 0.4155$; $R^2 = 0.0184$) and high (A: $y = 0.0626x + 0.2302$; $R^2 = 0.197$; B: $y = -0.0002x^2 + 0.021x + 0.465$; $R^2 = 0.1553$) levels of transformation (Figure 6.22). Neighbourhood transformation was measured as the percentage of the site and its surrounding sites (viz. 3x3 site rectangle) were mapped as transformed.

The significance of this third relationship is two fold. Firstly, land-value is a relatively good, albeit crude, indicator of conservation priority in landscapes where little other information is available on the biodiversity or land-use of the area. Areas with the highest land prices are where conservation action should be prioritized as options for achieving targets in these landscapes are diminishing rapidly. Secondly, the state and conservation authorities must be prepared to accept that achieving conservation targets in transformed landscapes is going to be exponentially more expensive than doing so in untransformed landscapes. In the Knersvlakte, land set aside for conservation in a priority area such as the Olifants River valley could cost 1000 times more than land only a few kilometers away from the river with no cropping or mining potential.

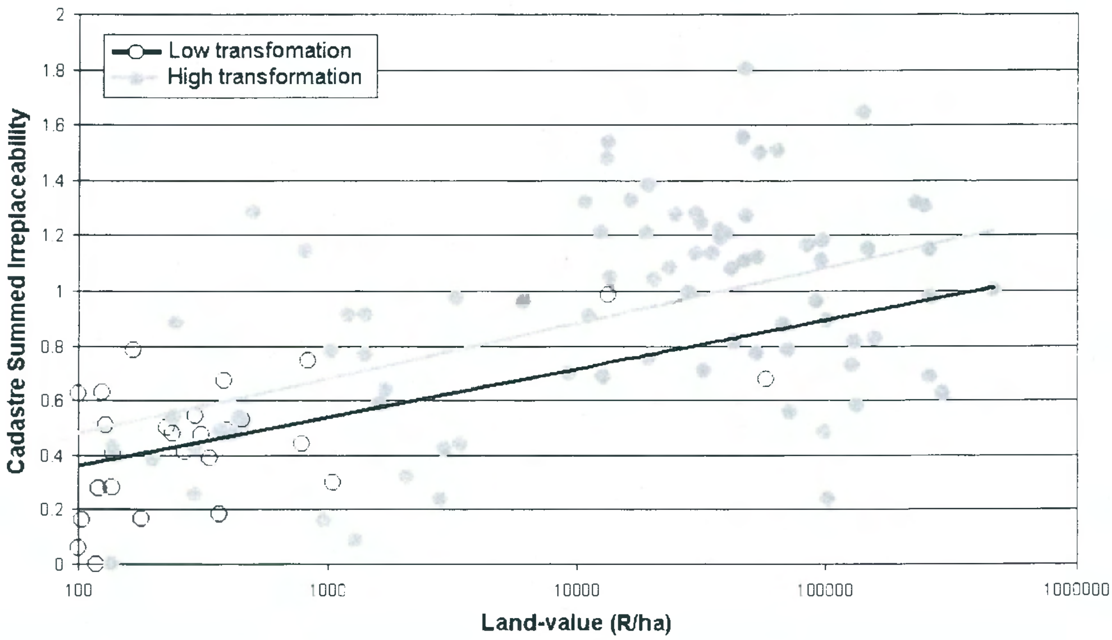


Figure 6.21: The relationship between conservation options (summed irreplaceability) and the value of land (price paid for properties in R/ha) in an area of low (y = 0.0773Ln(x) + 0.0055; R^2 = 0.2643) and high (y = 0.0874Ln(x) + 0.0788; R^2 = 0.2403) transformation in the planning domain (Figure 6.22). Summed irreplaceability was calculated as the average for all sites that covered any cadastre greater than 2ha sold between June 1998 and June 2001.

In landscape characterized by moderate to high levels of transformation, irreplaceability will always be driven by this transformation. In other words, priorities for land-use and conservation action will be dictated by where transformation is most extensive or happening at the fastest rate. At the other end of the spectrum, in landscapes with low levels of transformation irreplaceability will be determined by the distribution of biodiversity in the landscape. Those areas with the greatest number of features will invariably be the highest priority areas. What is most important to remember is that irreplaceability is not a measure of conservation value or priority. It is a measure of options to achieve a set of targets.

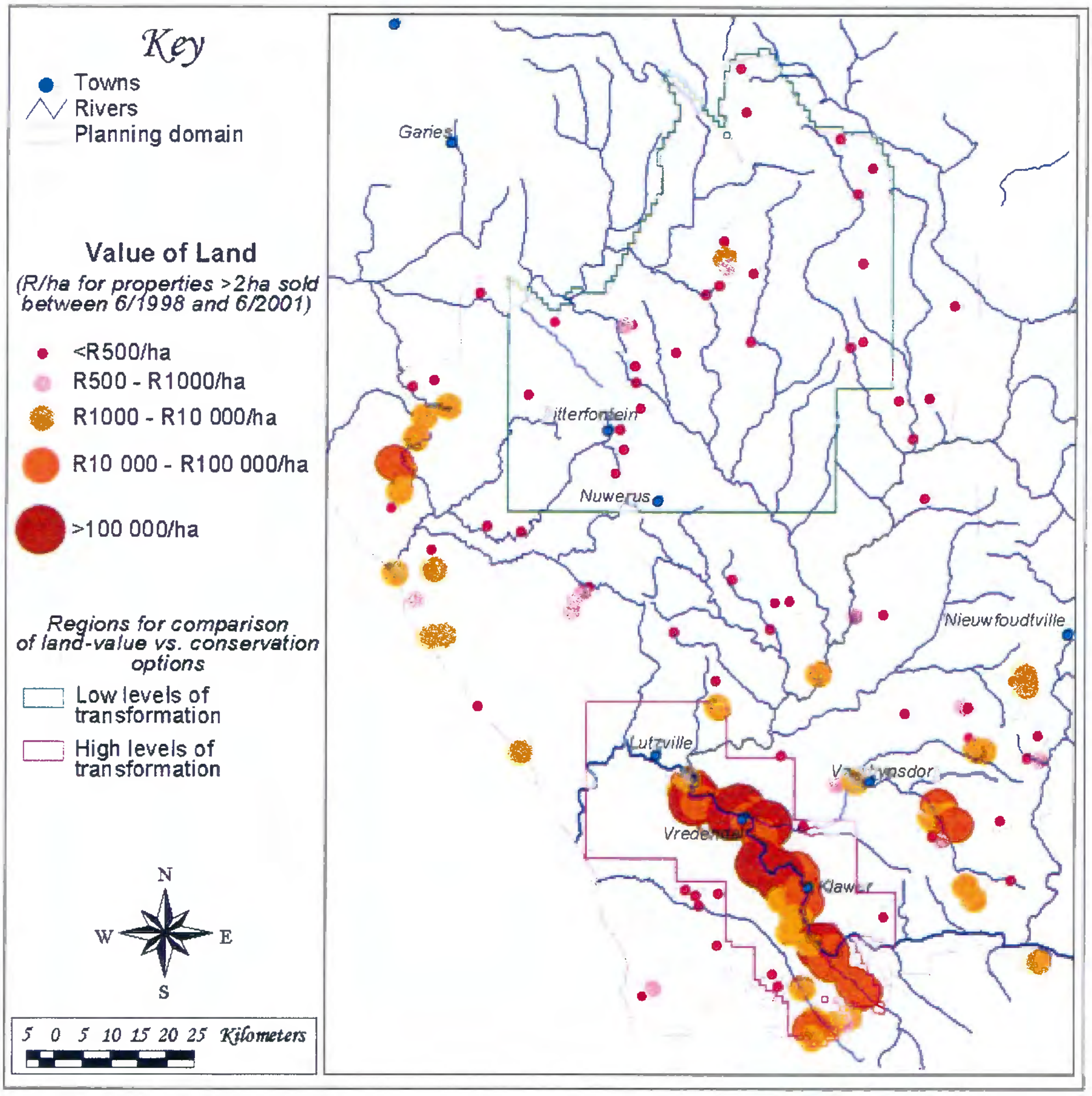


Figure 6.22: The value of land (R/ha) in the planning domain for properties greater than 2ha sold between June 1998 and June 2001. The results of the comparison between land-value and conservation options are presented in Figure 6.21.

6.6 Discussion

Through the design of the biosphere reserve it is estimated that 70% of the planning domain is required to represent all mapped biodiversity pattern features and maintain a minimum of ecological processes necessary for this biodiversity to persist. This is a significant chunk of land, but it is on a par with studies form elsewhere in the world that have assessed the spatial requirements for maintaining ecological processes (Chapter 4).

The biosphere reserve outline presented here should be regarded as a biocentric vision for the sustainable development of a landscape. It is not a plan for a biosphere reserve. The detail is too coarse for on the ground land-use zonation; there is no schedule of activities for implementation of the plan; and, not all human requirements have been adequately considered.

The priority setting exercise shows where implementation of this vision should act first. There is no net benefit to conservation in spending all the time and budget on creating a magnificent biosphere reserve in areas with low vulnerability, i.e. in areas where if there was no conservation action at all there would be little or no loss to conservation as no land-use beyond stock farming can be undertaken. Efforts should be focused on areas where if there were no immediate action the biodiversity targeted would be compromised due to transformation. Ultimately this would mean that conservation goals would not be met.

Where should implementation act first? Where options to achieve the biodiversity goals are retreating most rapidly. The immediate priorities for implementation of the biosphere reserve are the lower Olifants River and the coastal forelands. Should opportunities present themselves elsewhere in the planning domain then the implementation process should capitalize on these especially if they do not retract resources from the immediate priorities. For both the conservation and land-use planning authorities these two geographic areas are where immediate resources should be focused. Invariably, action will be correlated with transformation as this is where options for achieving targets has diminished most and where in some cases they are still retreating. Management of land-use in these landscapes will be a critical component of the success of the biosphere reserve.

In terms of action, for both the core and biosphere reserves this study only really discusses the most immediate priorities. Experience from the previous Knersvlakte study (Desmet *et al.* 1999) would indicate that at some point in the future priorities for action will be re-assessed based on the progress made to that date and updated patterns in land-use activities. Thus it does not make sense to draw up a schedule for action that will only be implemented in five or ten year's time. By this time changing land-use patterns may dictate that the priorities change to reflect changing human environment.

The boundaries identified for the two broad land-use categories are potentially flexible. By overlaying the final boundary map on the initial irreplaceability map will provide a degree of insight as to where there may be flexibility. Areas with high irreplaceability values are unlikely to have much flexibility, as there are few options for achieving the conservation targets elsewhere in the landscape. Areas where there are low irreplaceability values indicate where there may be a degree of flexibility in achieving targets. Added to this are the caveats such as missing features discussed in Chapters 5.2. Incorporating such omissions and newly identified features can only practically be addressed in successive planning exercises, but these will influence the flexibility of some areas identified presently as flexible.

The planning domain is a landscape of contrasts. For the vast majority of the area there is very low population density, and development or transformation of the landscape, grazing impacts excluded, is very low. In contrast to this quiet rural landscape is the densely developed lower Olifants River region and the extensively cultivated wheat-lands to the south and east. For the majority of biodiversity features it is possible to achieve their targets whilst accommodating existing and potential future development. In contrast, transformation, especially in the south, dictates that nearly all the remaining habitat in these areas is required to achieve the conservation targets.

In addition to transformation, the outcome of the planning process, i.e. the location of the two broad land-use categories, is constrained further by how biodiversity is distributed throughout the landscape. The initial irreplaceability maps indicate that almost all areas of the landscape are important for some or other component of biodiversity. The implications of this are two-fold. Firstly, no single statutory reserve will be able to fulfill the nations obligations to represent all components of biodiversity in a formal reserve. The reserve system in the Knersvlakte will need to comprise an extended network of both statutory and non-statutory conservation areas in order to achieve this goal. Secondly, accommodating ecological processes in planning is land-hungry and requires a broader view of the landscape. Land-use planning and management needs to adopt a landscape perspective and be aware of the implications of local actions in the broader landscape context.

This study has highlighted some important aspects of the planning process that should be developed in future such studies. These are how processes are included in the planning process; the role of experts and the current deficiencies in planning software; and, method for determining priorities for action.

There is a need to hard-wire ecological processes into the feature dataset. Within a C-Plan planning context this needs to be done if they are to be adequately considered during the planning and also the implementation assessment processes. Failing to do this means that it is very difficult to keep track of how well targets for these processes have been achieved. Current methods for including processes are inadequate as they constrain the reserve design process (see Section 6.3.2).

By using non-hardwired processes and design rules does allow for the inclusion of a range of spatially unfixed processes. This does mean that expert input forms an essential and integral part of the planning process when deciding which sites to include in the reserve network. Expert input is also essential for gathering input data. There is a limit to the availability and amount of biodiversity data that can be gathered during a project for inclusion in the feature-set. Expert input such as the SKEP expert area can be used to complement or extend available datasets.

One of the biggest limitations of the planning process is that the current planning software does not consider design or ecological function aspects of planning such as connectivity. Design here was purely a rule-moderated expert driven process to achieve a set of explicit targets. There are plans to incorporate a connectivity component into C-Plan via MARXAN (B. Pressey, pers. Comm.). This should help provide more explicit reasoning or why or where areas in the landscape are selected. Overall, though, there is no generic conservation planning software package that adequately considers biodiversity pattern and process targets as well as considering connectivity and which can be applied to any planning domain. If systematic conservation planning is to progress then a significant amount of time and money will have to be invested into developing decision support software that can assist practitioners in giving effect to the principles of systematic conservation planning.

The discussion on irreplaceability (see Section 6.5.4) highlights some interesting patterns in irreplaceability values as a result of transformation. Broadly speaking, priorities for action in any landscape will always be in areas where there is most transformation as these are areas where options are most constrained and where the transformation is ongoing this is where options are retreating fastest (Pressey 1994c; Richardson *et al.* 1996; Etter 2000; Wessels *et al.* 2003). In the Knersvlakte even there was no biodiversity data available at all it would have been possible to determine the same broad immediate priority areas for action simply using a transformation and current land price map.

Current land price is very useful information as it can be assumed that the market determines land price based on the current and future economic potential of that land. Whilst the alternative land-use maps used in this study provide only information on the potential for various land-uses, land price takes this one step further by providing a measure of where this potential is and will be realized in the short term. Applying this logic to areas without good biodiversity data might be a very good proxy for determining immediate priorities for conservation action in the absence of any biodiversity data. Certainly for South Africa, acquiring land price information is relatively straight forward as all land sales back to 1986 are electronically data-based and housed with the Deeds Office in Pretoria.

Both the core reserve and biosphere reserve design chapters have highlighted the importance of identifying and setting targets for biodiversity pattern and process. Whilst the species area relationship holds much promise for determining biologically meaningful biodiversity pattern targets, and the landscape ecology research does begin to make in roads into setting meaningful process targets, there is still a major challenge not adequately addressed in this study for spatially identifying and setting targets for processes. As the systematic conservation planning framework rests on explicit targets begin set for biodiversity features this task in effect proves to be the single most crucial step in implementing the planning protocol.

6.7 Appendices

Appendix 6.1: The complete feature table derived for planning the biosphere reserve. The target is percentage of original extent of feature.

	Feature ID	Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
1	KMB	Kamiesberg Mountain Brokenveld	24700.75	24170.25	60	14820.45
2	NK	Namaqualand Klipkoppe	217185.75	193479.25	60	130311.45
3	PQGP	Platbakkies Quartz and Gravel Patches	883.50	883.50	60	530.10
4	BB	Bushmanland Basin	3504.75	3504.75	60	2102.85
5	NK_F	Namaqualand Klipkoppe Flats	44591.25	43385.00	60	26754.75
6	NLSK	Namaqualand Lowland Succulent Karoo	19545.25	17753.00	60	11727.15
7	HK	Hantam Karoo	661.00	661.00	60	396.60
8	NAG	Namaqualand Arid Grasslands	45734.50	42963.75	60	27440.70
9	NKLSK	Northern Knersvlakte Lowland Succulent Karoo	116060.50	115987.75	60	69636.30
10	NQSK	Nuwerus Quartzite Succulent Karoo	62503.50	61481.00	60	37502.10
11	NA	Namaqualand Alluvia	8239.50	3898.50	60	4943.70
12	NRSP	Namaqualand Red Sand Plains	113094.25	90785.00	60	67856.55
13	NSF	Namaqualand Sand Fynbos	39369.50	37489.00	60	23621.70
14	KNEQP	Knersvlakte Quartzfields	122444.00	121247.00	60	73466.40
15	VSR	Vanrhynsdorp Shale Renosterveld	34832.00	33954.00	60	20899.20
16	KOTQP	Kotzerus Quartz Patches	365.00	365.00	60	219.00
17	KS	Knersvlakte Shales	71978.00	71844.75	60	43186.80
18	KOMQP	Komkans Quartz Patches	27330.25	27265.00	60	16398.15
19	NSS	Namaqualand Southern Strandveld	1834.00	1793.00	60	1100.40
20	SKLSK	Southern Knersvlakte Lowland Succulent Karoo	98306.50	81558.00	60	58983.90
21	BSF	Bokkeveld Sand Fynbos	58872.25	37061.00	60	35323.35
22	CKLSK	Central Knersvlakte Lowland Succulent Karoo	16768.50	16652.25	60	10061.10
23	ACSM	Arid Coastal Salt Marshes	3433.50	2627.25	60	2060.10
24	RQSK	Rooiberg Quartzite Succulent Karoo	16598.00	16473.75	60	9958.80
25	NSG	Namaqualand Spinescent Grasslands	49487.75	47666.25	60	29692.65
26	KOEQP	Koekenaap Quartz Patches	1599.25	1549.00	60	959.55
27	ORQP	Olifants River Quartz Patches	21557.25	10221.75	60	12934.35
28	KD	Knersvlakte Dolorites	2640.25	2558.00	60	1584.15
29	TTRQP	Troe-Troe River Quartz Patches	5017.50	5015.00	60	3010.50
30	LBF	Lamberts Bay Strandveld	38099.00	29986.50	60	22859.40
31	LSF	Leipoldtville Sand Fynbos	51061.25	24007.75	60	30636.75
32	DRSK	Doring River Succulent Karoo	12073.25	8123.25	60	7243.95
33	RQP	Remhoogte Quartz Patches	3338.75	3135.50	60	2003.25
34	ASSK	Agter-Sederberg Succulent Karoo	979.50	438.75	60	587.70

	Feature ID	Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
35	GSF	Graafwater Sandstone Fynbos	3937.25	2611.50	60	2362.35
36	LC_1	Land Class 1	2.00	2.00	60	0.00
37	LC_2	Land Class 2	90.00	76.25	60	45.75
38	LC_3	Land Class 3	100185.00	99399.25	60	59639.55
39	LC_4	Land Class 4	16636.50	14143.50	60	8486.10
40	LC_5	Land Class 5	1062.00	604.75	60	362.85
41	LC_6	Land Class 6	16484.00	15364.00	60	9218.40
42	LC_7	Land Class 7	96340.75	88948.00	60	53368.80
43	LC_8	Land Class 8	19347.75	18366.75	60	11020.05
44	LC_9	Land Class 9	227.00	226.00	60	135.60
45	LC_10	Land Class 10	989.00	989.00	60	593.40
46	LC_11	Land Class 11	14.00	14.00	60	0.00
47	LC_12	Land Class 12	2429.00	2195.75	60	1317.45
48	LC_13	Land Class 13	49380.50	48761.00	60	29256.60
49	LC_14	Land Class 14	34.00	34.00	60	0.00
50	LC_15	Land Class 15	4.00	4.00	60	0.00
51	LC_16	Land Class 16	864.00	854.00	60	512.40
52	LC_17	Land Class 17	1485.00	1475.50	60	885.30
53	LC_18	Land Class 18	499.00	430.25	60	258.15
54	LC_19	Land Class 19	5239.00	5216.00	60	3129.60
55	LC_20	Land Class 20	3.00	3.00	60	0.00
56	LC_21	Land Class 21	87639.75	83373.75	60	50024.25
57	LC_22	Land Class 22	133538.25	92219.00	60	55331.40
58	LC_23	Land Class 23	2.00	2.00	60	0.00
59	LC_24	Land Class 24	1646.75	1215.75	60	729.45
60	LC_26	Land Class 26	5642.00	5639.00	60	3383.40
61	LC_27	Land Class 27	26829.25	26355.50	60	15813.30
62	LC_28	Land Class 28	461.00	454.50	60	272.70
63	LC_29	Land Class 29	45920.50	45375.25	60	27225.15
64	LC_30	Land Class 30	40961.50	38695.00	60	23217.00
65	LC_31	Land Class 31	67553.00	63651.00	60	38190.60
66	LC_32	Land Class 32	76224.50	65873.00	60	39523.80
67	LC_33	Land Class 33	741.00	737.00	60	442.20
68	LC_34	Land Class 34	20871.25	16497.50	60	9898.50
69	LC_35	Land Class 35	180.00	180.00	60	108.00
70	LC_37	Land Class 37	1.00	1.00	60	0.00
71	LC_38	Land Class 38	1610.00	1544.75	60	926.85
72	LC_39	Land Class 39	40836.75	26811.25	60	16086.75
73	LC_40	Land Class 40	75883.75	70131.75	60	42079.05
74	LC_41	Land Class 41	51.00	51.00	60	30.60
75	LC_42	Land Class 42	2030.00	1505.75	60	903.45
76	LC_43	Land Class 43	4923.00	4666.50	60	2799.90
77	LC_44	Land Class 44	299.00	294.00	60	176.40
78	LC_45	Land Class 45	104.75	104.75	60	62.85
79	LC_46	Land Class 46	56208.75	45857.00	60	27514.20
80	LC_47	Land Class 47	48463.00	44523.75	60	26714.25
81	LC_48	Land Class 48	71612.00	64071.75	60	38443.05
82	LC_49	Land Class 49	2446.75	2397.25	60	1438.35

Feature ID		Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
83	LC_50	Land Class 50	19940.00	18836.75	60	11302.05
84	LC_51	Land Class 51	39.00	39.00	60	0.00
85	LC_52	Land Class 52	26309.00	23222.50	60	13933.50
86	LC_53	Land Class 53	9.00	9.00	60	0.00
87	LC_54	Land Class 54	10394.00	9542.75	60	5725.65
88	LC_55	Land Class 55	1433.00	1298.00	60	778.80
89	LC_56	Land Class 56	418.25	418.25	60	250.95
90	LC_59	Land Class 59	1845.00	1740.00	60	1044.00
91	LC_61	Land Class 61	1445.25	1442.75	60	865.65
92	LC_63	Land Class 63	87089.00	76814.50	60	46088.70
93	LC_64	Land Class 64	10389.00	10082.25	60	6049.35
94	LC_65	Land Class 65	76.00	75.50	60	45.30
95	LC_66	Land Class 66	14506.00	10084.00	60	6050.40
96	LC_67	Land Class 67	654.00	596.75	60	358.05
97	LC_69	Land Class 69	28.00	28.00	60	0.00
98	LC_70	Land Class 70	58.00	58.00	60	34.80
99	LC_73	Land Class 73	6222.75	2407.00	60	1444.20
100	LC_74	Land Class 74	688.50	652.50	60	391.50
101	LC_76	Land Class 76	7831.00	7624.50	60	4574.70
102	LC_77	Land Class 77	873.00	867.75	60	520.65
103	LC_78	Land Class 78	378.00	377.00	60	226.20
104	LC_79	Land Class 79	802.00	802.00	60	481.20
105	LC_80	Land Class 80	7672.00	6348.00	60	3808.80
106	LC_82	Land Class 82	7708.00	5708.75	60	3425.25
107	LC_83	Land Class 83	11.00	11.00	60	0.00
108	LC_84	Land Class 84	8.00	8.00	60	0.00
109	LC_85	Land Class 85	129.00	129.00	60	77.40
110	LC_86	Land Class 86	4726.00	2147.00	60	1288.20
111	LC_91	Land Class 91	58.00	58.00	60	34.80
112	LC_94	Land Class 94	7.00	7.00	60	0.00
113	LC_97	Land Class 97	5.00	5.00	60	0.00
114	LC_99	Land Class 99	3.00	3.00	60	0.00
115	EXP_13	Plant ms6	1.00	1.00	100	1.00
116	EXP_15	Invert JI33 Kommandokraal	16.00	16.00	100	16.00
117	EXP_4	Plant pgd22	291.00	291.00	12	34.92
118	EXP_17	Fish f4	1257.25	1257.25	100	1257.25
119	EXP_14	Invert JI32 Marble hills	1643.25	1643.25	100	1643.25
120	EXP_16	Fish f2	3108.25	3108.25	50	1554.13
121	EXP_1	Plant pgd41	3737.50	3737.50	100	3737.50
122	EXP_2	Plant pgd18	4784.75	4784.75	75	3588.56
123	EXP_12	Plant ms4	5650.50	5650.50	75	4237.88
124	EXP_11	Plant ms5	5774.00	5774.00	75	4330.50
125	EXP_19	Birds b22	8578.75	8578.75	50	4289.38
126	EXP_9	Plant pgd16	10033.00	10033.00	50	5016.50
127	EXP_18	Fish f3	10206.00	10206.00	50	5103.00
128	EXP_8	Plant pgd17	11311.75	11311.75	50	5655.88
129	EXP_10	Plant ms3	13584.75	13584.75	50	6792.38
130	EXP_7	Plant pgd19	32597.25	32597.25	12	3911.67

Feature ID	Feature Name	Original Extent of Feature (ha)	Area Available (ha)	Target (%)	Target (ha)
131 EXP_6	Plant pgd15	33757.75	33757.75	12	4050.93
132 EXP_5	Plant pgd20	36283.00	36283.00	12	4353.96
133 EXP_20	Amphibian ao7	126787.50	126787.50	10	12678.75
134 EXP_3	Plant pgd14	165443.50	165443.50	10	16544.35
135 EXP_21	Amphibian ao6	277081.50	277081.50	10	27708.15
136 SEA_FACING	Sea-facing topographic climate refugia	62835.25	62835.25	90	5655172.50
137 S_SLOPES	South-facing slopes climate refugia	98164.00	98164.00	80	7853120.00
138 TOPO_REF	Topographic climate refugia	225024.50	225024.50	70	15751715.00
139 EDPH_INTER	Edapic interfaces (buffer 500m)	311990.50	311990.50	80	24959240.00
140 RIVER_50	River Order 1 (buffer 50m)	10088.39	10088.39	80	807070.88
141 RIVER_100	River Order 2 (buffer 100m)	5906.83	5906.83	80	472546.00
142 RIVER_200	River Order 3 (buffer 200m)	7326.52	7326.52	80	586121.76
143 RIVER_250	River Order 4 (buffer 250m)	5681.16	5681.16	80	454492.96
144 RIVER_500	River Order 5 (buffer 500m)	13479.80	13479.80	80	1078383.60
145 VIEW_SH	Viewshed	629083.75	629083.75	95	597629.56
146 FH_1	W quartz patches	2263.25	2263.25	60	1357.95
147 FH_2	W intermediate heuweltjie/quartz veld	1252.50	1252.50	60	751.50
148 FH_3	N quartz patches	11383.50	11383.50	60	6830.10
149 FH_4	N intermediate heuweltjie/quartz veld	5250.50	5250.50	60	3150.30
150 FH_5	Central quartz patches	13130.00	13130.00	60	7878.00
151 FH_6	Central intermediate heuweltjie/quartz veld	5345.75	5345.75	60	3207.45
152 FH_7	Limestone	4920.75	4920.75	60	2952.45
153 FH_8	SE intermediate heuweltjie/quartz veld	757.25	757.25	60	454.35
154 FH_9	SE quartz patches	1267.75	1267.75	60	760.65
155 FH_10	SW quartz patches	421.75	421.75	60	253.05
156 FH_11	SW intermediate heuweltjie/quartz veld	32.50	32.50	60	19.50
157 FH_BUFF	Quart-patch areas	191155.91	191155.91	60	114693.54

7 Conclusions

Biodiversity surrogates are necessary to represent true biodiversity in conservation plans. In Chapter 2 lessons learnt form using a range of biodiversity surrogates in conservation plans were summarized by a set of rules for incorporating biodiversity surrogates into regional conservation assessments. These rules could prove useful for conservation planning in general:

Rule 1: There are no perfect biodiversity surrogates. Different surrogates focus on different aspects of biodiversity so expect different results.

Rule 2: Use as many surrogates in a conservation plan as are available within the projects time and budget constraints.

Rule 3: As a minimum, the primary biodiversity surrogate layer should be continuous surrogate that covers the entire planning domain.

Rule 4: Do not under estimate the contribution that spatially explicit expert derived data can make to the planning process.

Rule 5: Do not discard incomplete point taxonomic datasets, as it is better to base decisions on what you know rather than what you don't know.

Rule 6: QDS scale taxonomic data are not useful for on the ground conservation decision-making.

Certainly a guideline for the custodians of biodiversity data in South Africa is that they have to seriously re-think how they collect and archive biodiversity data. Despite the massive amount of biodiversity information available in this country the fact that it is predominately at the QDS scale renders it functionally useless for on the ground conservation planning and action. We need to address this problem as a matter of urgency in this country.

The targets work in Chapter 3 is probably the most exciting aspect of this thesis. I developed the present formulation the method by dredging through some of the early work on the SAR contained in key texts on the subject such as Rosenzweig (1995) and Diamond and May (1976). I have subsequently found that the SAR has been used in a range of applications in conservation. Pressey *et al.* (2003b) used the SAR to set differential targets for parts of the of the Cape Floristic Region and made recommendations for its application at the scale of vegetation type. Vreugdenhil *et al.* (2003) have also proposed using the SAR to set representation targets. The SAR is also being used in systematic conservation planning in Australia (Simon Ferrier and Dan Faith, pers. comm.) However, to the best of my knowledge no one has ever used it quite so explicitly, using real data, to set targets for biodiversity features in a conservation plan. This method could revolutionise target setting in conservation planning. For the first time we have a method for setting representation targets that is based on ecological theory and uses real data. It gives us an ecological basis on which to defend the targets we set for conservation, something the "10% rule" never did. The drawback with this method is that systematic inventory data (e.g. phytosociological relevés) are lacking for much of the world.

The method is not without its problems. A further drawback lies in the method for estimating the true species richness of a land-class. Whilst EstimateS is a great piece of software, the estimators are still only statistical models for making this prediction and there is certainly no consensus as to which is the better one for the job (Gotelli and Colwell 2001). I am currently working on a method, in conjunction with Simon Ferrier and Dan Faith, which attempts to estimate the z-value without the need to estimate the true species richness. This would certainly remove some uncertainty from the whole equation.

I think the discussion on process or landscape targets is equally very exciting. There is a wealth of information and ideas in other fields of ecology that can prove to be potentially useful in conservation planning. Chapter 4 stresses, firstly, that conservation planning is an integrative discipline that draws on all fields of ecology. Secondly, and more importantly, the size of areas required to avert extinction discussed in this chapter are far greater than the conservation targets developed to merely represent biodiversity.

There is an ecologically justifiable, if somewhat tenuous, basis for setting a ballpark 60% landscape target for conservation of ecological processes. The implications of this "rule" for conservation and more broadly land-use planning are significant. Firstly, we can never hope to adequately conserve biodiversity within the formal reserve network. We may be able to fully represent biodiversity in a reserve network that covers 20% of a landscape, but for this biodiversity to persist the reserve network is inextricably linked to processes in the surrounding matrix. Secondly, it follows that conservation planning should not be planning for a reserve network, but for land-use across the entire landscape where conservation is only one of several possible land-use options. There is a need in conservation sectors to dispel the notion of dealing only with reserves. There needs to be a paradigm shift in conservation from an on/off reserve mindset to a more integrative whole landscape mindset.

The first three data chapters of this thesis dealt with some issues around the biodiversity data that we use in planning and also setting targets for these biodiversity features. The last two chapters provide an example of how these data can be used in systematic conservation plans to address two different kinds of conservation planning questions.

Chapter 5 looks at how one can use the systematic planning approach to design a statutory reserve that meets a set of criteria, in this case a reserve that represents the unique biodiversity attributes of the planning domain – the quartz patches, etc. This plan was done for the implementing agency to help guide them as to which properties to include in the reserve and how to schedule implementation, i.e. a typical reserve design type scenario. Chapter 6 takes the same biodiversity data but applies it to a different scale of conservation question – the design of a biosphere reserve. This application in effect begins to make the transition between reserve-focused conservation planning and landscape-level, biocentric land-use planning. This is where the future of conservation planning lies. Both approaches, however, use the same six-point planning protocols that make them "systematic plans".

There are some key gaps in these plans that should be addressed at some stage.

Firstly, the issue of assessing vulnerability is not dealt with adequately. This is a whole field of study in its own right that is probably more suited to an economist than a biologist. When it comes to implementation information on current land-use and projected future land-use are probably more important than the biodiversity information in determining priorities for action. This is certainly a field where conservation planning has to improve. I would predict that in the future the biodiversity component of systematic plans would actually form a very small component of the overall information and analyses that go into developing a plan. At the end of Chapter 6 I demonstrate that it is possible to make relatively accurate decisions about conservation priorities in the absence of any biodiversity data and using only a crude measure of landscape transformation. No one can argue that they cannot do a conservation plan because there are no biodiversity data!

Secondly, the manner in which the landscape level ecological processes are operationalised needs to be addressed. I do not agree entirely with, for example, the CAPE (Rouget *et al.* 2003b) and STEP (Cowling *et al.* 2003a) methods of hard wiring landscape level processes into the feature dataset as this removes some flexibility from the planning process. The rule-based approach that is used here allows for greater flexibility, but it does not allow one to keep track of the target achieved. How ecological processes are operationalised in conservation plans requires significant work. Maybe we were constrained by the limitations of the planning software, viz. C-Plan, and perhaps the marriage between C-Plan and Marxsan will open a whole new chapter on how we consider ecological processes in planning. I think the biggest stumbling block, though, is that we do not adequately understand how the majority of ecological processes operate in space. Owing to our ignorance of processes we could retain 60% of a landscape and still standby and watch the natural fabric of a landscape unravel.

Overall, there are three broad conclusions that I can draw from the research presented in this thesis. These are:

- Systematic conservation planning is essential if one's goal is to save biodiversity. Biodiversity is a complex entity than cannot be easily summarised in a few simple indices. Ad-hoc planning that attempts to consider all aspects of biodiversity outside of a systematic planning framework will eventually fail. Some aspects of biodiversity will eventually fall through the gaps, as one cannot keep track of the fate of all aspects of biodiversity in an ad-hoc manner. Most importantly, systematic approaches promote transparent decision-making allowing a broader spectrum of role players to participate in the process. They also make explicit what we are trying to achieve so at least everyone knows where we are heading and what we would like to achieve.

- From my experiences here I can conclude that setting targets is the most challenging stage in the systematic planning protocol. Targets are where one sets out what a conservation plan is going to achieve. They also provide the measure against which the success or failure of your plan will be assessed. All subsequent stages in a systematic plan development and implementation are dependent on the targets that you set. They are perhaps the single most important aspect of a systematic conservation plan. Traditionally, however, targets have been clouded by the debate as to their biological basis or relevance. I demonstrate here that it is possible to derive biologically meaningful conservation targets for both biodiversity pattern and potentially processes. What I present here represents a first stab at the problem and there is still much work to be done here.

- If you want to save biodiversity then you need to consider the whole landscape and not simply the formal reserve network. Like human land-use planning makes decision about all parts of the landscape, so conservation planning needs to explicitly consider the fate of biodiversity in every parcel of land, even transformed areas. The discussion on landscape functionality targets and the Knersvlakte biosphere reserve plan demonstrate the need for this approach quite clearly. "Biodiversity", "the landscape", "an ecosystem" or however one chooses to conceptualise the living world that we are trying to conserve can be viewed as a giant organism. Conservation planning is really about figuring out how much of this organism we can consume before it dies. Planning only to conserve the brain or heart of the organism in isolation is not a cleaver strategy for ensuring the organism's longterm survival. In addition to considering biodiversity patterns and processes, an integrative landscape level approach to conservation planning is essential if one's goal is to maintain living landscapes.

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