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Measuring conservation value at fine and broad scales: implications for a diverse and fragmented region, the Agulhas Plain

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Abstract

This study explores the implications of spatial scale for conservation planning in the Agulhas Plain, South Africa. Regional planning relies on broad-scale data but fine-scale data are usually required for implementation at local level. This study addresses the implications of broad-scale planning for fine-scale implementation. Two original systems of notional reserves were developed for this region using C-plan, a decision support system for systematic conservation planning. The first conservation plan was derived using broad scale data (1:250,000) and consisted of nine broad habitat units (land classes based on topography, geology, and climate), remote sensing mapping of habitat transformation and large planning units defined by 1/16th degree squares (average size 3900 ha). The second system was identified at a finer scale (1:10,000) using 36 vegetation types (mapped in the field), ground survey mapping of habitat transformation and cadastral boundaries as planning units (average size 252 ha). Using classification trees, this study compared reserve-design efficiency (the area required to achieve conservation targets), the spatial patterns of conservation value (the irreplaceability value of planning units), biodiversity features, and habitat transformation at both scales. A similar amount of land was required to meet all conservation targets (identified using minimum set analysis) at the broad and fine scale. There was considerable overlap between the two conservation plans as most of fine-scale conservation targets could be achieved under the broad-scale conservation plan. However, irreplaceability values, which measure the likelihood of selecting planning units for achieving representation targets, were much higher at the fine scale. The use of broad-scale biodiversity features underestimated irreplaceability value at a fine scale in heterogeneous and fragmented portions of the landscape. The implications of moving from broad- to fine-scale conservation planning, as well as their respective benefits are discussed. Maximising biodiversity conservation while minimising cost and resources might be achieved by a combination of broad-scale assessments for relatively homogeneous and untransformed areas and fine-scale ones for heterogeneous and fragmented areas.

Keywords: Biodiversity mapping; Conservation planning; Habitat transformation; Reserve design; Reserve-selection algorithm; Spatial scale

1. Introduction

In past decades, biodiversity was perceived largely in terms of species richness, and conservation attention was often directed at hotspots rich in total species or rare species (Noss, 1987). More recently, there has been a shift in conservation planning, and protection strategies are increasingly based on broad-scale approaches, conserving biodiversity at the ecosystem level across whole regions (Franklin, 1993; Mittermeier et al., 1998; Soulé and Terborgh, 1999; Schwartz, 1999; Poiani et al., 2000). This emphasis on broad-scale conservation is highlighted by recent ecosystem-based management policies by federal agencies in the USA (Christensen et al., 1996; Soulé and Terborgh, 1999). An important aspect of this approach is the use of a "coarse-filter" of targeted features (Noss, 1987) such as communities, habitats, landscapes, ecosystems or more recently focal species (Lambeck, 1997) to serve as surrogates for the distribution of biodiversity.

Although broad-scale conservation has several advantages, such as the preservation of ecosystem linkages and processes and the incorporation of a large proportion of associated species, its implementation is not straightforward. There is no a priori basis for prioritising the ecosystem attributes to form the classification of coarse filter entities (e.g. vegetation types, environmental units), and various methods have been

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suggested (Heywood, 1995). There is no accepted classification system for communities or ecosystems and questions remain about appropriate levels of the classification hierarchy for recognition, inventory and protection of natural communities (Ferrier, 2002).

The emergence of remote-sensing systems and Geographic Information System procedures for identifying and measuring habitat structure or landscape types provides important opportunities for conservation planning, more particularly for coarse-filter conservation (Davis, 1995; Burke, 2000). The spatial scale at which conservation decisions are taken is, however, a crucial issue. The spatial resolution of data collection, habitat classification, or the size of planning units (used as the building blocks for systems of conservation areas) can greatly affect the outcomes of conservation planning (Pressey and Logan, 1998; Fuller et al., 1998). Information collected at one scale might not be appropriate to answer management questions at another scale (Rouget and Richardson, in press). The spatial resolution used in the habitat classification (i.e. the biodiversity features) will depend on the level of detail desired, but the choice of resolution is generally determined and constrained by the availability of data and resources (mainly time and finance). So, although fine-scale data on biodiversity features such as vegetation types is desirable due to their greater content of information on biophysical variation (Beckett and Burrough, 1971; Rowe and Sheard, 1981; Pressey and Bedward, 1991), maps of biodiversity features for large regions are generally produced at a coarse resolution.

The constraints on fine-scale mapping across large regions introduce a problem for conservation planning. Regional assessments of conservation priority are important to place individual areas in the context of natural processes and interactions. Consistent data on coarse-filter biodiversity features across whole regions are usually available only at broad scales (e.g. 1:1,000,000 or 1:250,000). But conservation plans produced at these scales must be implemented at finer scales, often below the resolution of the regional data sets. In some parts of regions, there will be localised, fine-scale data sets available for implementation. In these situations, planners and managers will have to consider whether the assessment of conservation values derived in the regional plan with broad-scale data matches the assessment of values that would emerge from analysis of the fine-scale data. They might also have to consider replacing the values from the big picture with more localised ones based on the fine-scale biodiversity features. This transition from broad-scale planning to fine-scale implementation has received little attention.

The Cape Action Plan for the Environment (CAPE), a systematic conservation and implementation process for South Africa's Cape Floristic Region (CFR) (see Cowling and Pressey, 2003; Younge and Fowkes, 2003) provided a rare opportunity for assessing the effects of spatial scale on systematic conservation planning. Cowling and Heijnis (2001) mapped broad habitat units at 1:250,000 to support the CAPE process. In a small part of the southern CFR, finer-scale mapping of vegetation types at 1:10,000 and development of a conservation plan for achieving targets for these types were completed to provide insights for implementing conservation action on the ground (Cole et al., 2000). Between the broad- and fine-scale data sets, biodiversity features differed in terms of relative extent and minimum mapping unit, habitat transformation mapping differed in terms of minimum mapping unit, and planning units differs in terms of size.

This paper applies a common planning protocol to both data sets and compares the patterns of conservation values that emerge. Several measures have been suggested to quantify the conservation value of an area and to assess the efficiency of reserve systems. Conservation value of an area, and therefore its prioritisation, can be quantified by its likelihood of being required to achieve the set of targets for biodiversity features in a region, a measure termed its irreplaceability (Ferrier et al., 2000). Specifically, this study has three aims: (i) to compare the efficiency of achieving conservation targets at broad and fine scales; (ii) to compare differences in patterns of irreplaceability values at broad and fine scales; and (iii) to assess how the spatial scale of primary data layers (broad habitat units, vegetation types, and maps of transformation of native vegetation) affect the outcomes of conservation planning.

2. Methods

2.1. Study area

The Agulhas Plain (2160 km^2) is part of the Cape Floristic Region (CFR), one of the global hotspots of plant diversity and endemism (Myers et al., 2000). The Agulhas Plain is a low coastal peneplain with a complex mosaic of edaphic types and has a typical mediterranean-type climate (mean annual rainfall of 500 mm). It is home to at least 1751 vascular plant species. Most local endemic species (ca. 100) are edaphic specialists, occurring in small and scattered populations (Cowling and Holmes, 1992). The predominant vegetation types on the Agulhas Plain are fynbos (on nutrient-poor soils) and renosterveld (on more fertile soils), both sclerophyllous, fire-prone shrublands (Cowling, 1992). The area is extensively fragmented by agricultural practices, and some 40% of the original natural vegetation has already been transformed (Cole et al., 2000). Recently, the South African National Parks Board initiated the establishment of a national park to conserve the lowland fynbos and wetland ecosystems of the southern Agulhas

Plain (Heydenrych et al., 1999). The park configuration largely follows a previous reserve system identified by Lombard et al. (1997).

2.2. Study design

The CAPE and the Agulhas projects designed a network of notional reserves for the Cape Floristic Region (CFR), and the Agulhas plain, respectively. Although both projects shared the same procedure, they were developed independently, using separate data sets (see Table 1). The CAPE data set consisted of broad-scale data for biodiversity features (broad habitat units, BHUs), habitat transformation mapping-based on remote sensing-, and large planning units (1/16th degree squares). The Agulhas data set consisted of fine-scale data for biodiversity features (vegetation types), habitat transformation mapping-based on ground surveys-, and small planning units (cadastral units). Minimum sets and irreplaceability patterns were derived using C-Plan for the CAPE and the Agulhas data. As expected, differences in minimum set and irreplaceability patterns (see later) between fine and broad scale conservation planning emerged. These differences could result from three factors:

- the use of different biodiversity features (BHUs vs. vegetation types),
- the use of different methods for mapping habitat transformation (remote sensing vs. ground surveys), and
- the use of different planning units (sixteenth degree squares vs. cadastral units).

These differences could translate into inadequate implementation priorities when moving from the broad to fine scales. For example, the broad-scale assessment could fail to select high-priority areas that would only be identified using fine-scale data. This paper investigates the reasons for differences between minimum sets and irreplaceability values arising from broad and fine-scale conservation planning.

Because these differences could emerge from three factors, each factor (biodiversity features, habitat transformation mapping, and planning units) has to be analysed separately to assess their relative importance. Three derived data sets were therefore compiled that combine fine-scale data from the Agulhas project with one broad-scale component derived from the CAPE project (see "derived" data sets in Table 1):

- Derived data set 1 assessed the role of biodiversity features. It consisted of fine-scale mapping of habitat transformation (from ground-survey) and small planning units (cadastral units) but used broad-scale biodiversity data (nine BHUs).
- Derived data set 2 assessed the role of habitat transformation mapping. It consisted of fine-scale biodiversity features (36 vegetation types) and small planning units, but used broad-scale mapping of habitat transformation (from remote-sensing).
- Derived data set 3 assessed the role of planning units. It consisted of fine-scale biodiversity features (36 vegetation types) and fine-scale mapping of habitat transformation (from ground survey) but used large planning units (sixteenth degree squares).

Minimum sets and irreplaceability values obtained from these three derived data sets were compared with the original fine-scale data, the Agulhas data set (Table 1).

Broad-scale and fine-scale data are described below. Methods and details are provided by Cole et al. (2000) for Agulhas (fine-scale) and by Cowling et al. (1999b) for the Cape Floristic Region (broad-scale).

2.2.1. Biodiversity features

A system of land classes was developed to act as broad-scale biodiversity surrogates (1:250,000) for the Cape Floristic Region (CFR). Although floristic and botanical knowledge is substantial for the CFR, no

Table 1
Framework for analysing the effect of spatial scale on conservation planning

Data sets	Factor analysed	Biodiversity features	Habitat transformation	Planning units
Original				
Agulhas		Vegetation types (36)	Ground survey	Cadastral units
CAPE		BHUs (9)	Remote sensing	1/16th degree square
Derived				
Data set no 1	Biodiversity	**BHUs**	Ground survey	Cadastral units
Data set no 2	Habitat transformation	Vegetation types	**Remote sensing**	Cadastral units
Data set no 3	Planning units	Vegetation types	Ground survey	**1/16th degree square**

In the derived data sets, each factor shown in bold (biodiversity features, habitat transformation, and planning units) was taken from the broad-scale data set (CAPE) while the remaining two factors were taken from the fine-scale data set (Agulhas). Results from derived data sets 1, 2, and 3 were compared with results from the original Agulhas data set.

previous system of land classes at a sufficiently fine scale was available for the entire area that could be used for regional conservation planning (Cowling and Heijnis, 2001). In the CFR, plant biodiversity is largely driven by climate, geology and topography (Cowling, 1992). Therefore land classes were identified by Cowling and Heijnis (2001) on the basis of these three factors as well as expert knowledge. They derived broad habitat units (BHUs) using unique combinations of geology, climate and topography. One hundred and two BHUs were derived for the CFR planning domain, of which nine occur on the Agulhas Plain (Fig. 1a).

Fine-scale biodiversity surrogates were developed by mapping vegetation types in the field for the entire Agulhas Plain (2160 km^2). Vegetation units identified by Cowling et al. (1988) were used as the basis for vegetation groups for the study area. These vegetation units were further divided during field mapping by an expert botanist. Vegetation types were defined on the basis of dominant and differential plant species. Thirty-six vegetation types were mapped and these were used as biodiversity features for fine-scale (1:10,000) conservation planning (Fig. 1b). The minimum mapping unit was 1 ha for the fine scale (and 25 ha for the broad scale).

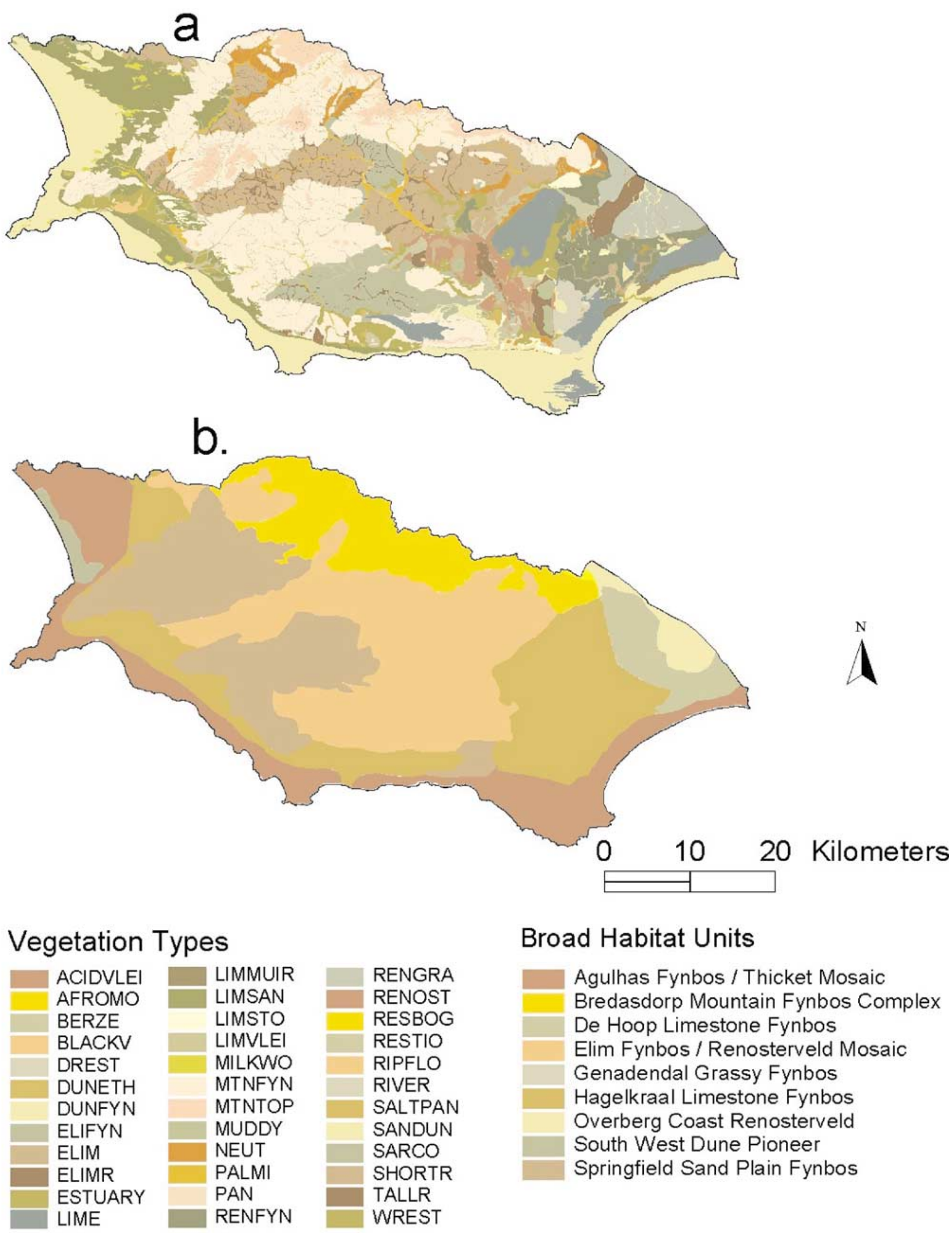

Fig. 1. Differences in biodiversity pattern arising from the scale of mapping biodiversity features. (a) Thirty-six vegetation types (1:10,000) were identified at fine-scale (Agulhas data set); (b) nine broad habitat units (1:250,000) were used for broad-scale conservation planning (CAPE data set).

2.2.2. Habitat transformation

Assessing the spatial extent and configuration of transformed areas (i.e. agriculture, urbanisation, and invasive alien plants) is crucial for conservation planning. The spatial dimensions of habitat transformation identify the area available for planning (i.e. area still considered as "natural") and play a role in setting conservation targets for biodiversity features (Pressey et al., 2003; see later). In both the broad- and fine-scale studies, areas covered by agriculture, urban development, and dense stands of invasive species (wattles, pines, and eucalyptus) were mapped as transformed and were not considered for achieving conservation targets. Current (1996) patterns of habitat transformation were assessed at broad scale for the entire CFR by means of satellite imagery (Lloyd et al., 1999; Rouget et al., 2003). Because of the scope of the CAPE project, the minimum mapping unit was 25 ha (i.e. no patch of natural vegetation was distinguished below 25 ha).

For the Agulhas Plain, fine-scale mapping of habitat transformation was derived from interpretations of aerial photographs (1:10,000) and complemented by extensive, expert-based ground survey (Cole et al., 2000). Most of the remaining vegetation occurs on small fragments. The minimum mapping unit was 1 ha.

2.2.3. Planning units

Planning units (also called selection units) consist of a priori subdivisions of the landscape and are used as the building blocks for systems of reserves (Pressey and Logan, 1998). They are usually different in size and configuration from the biodiversity features targeted for protection. Reserve selection algorithms, such as those embedded in C-Plan (Ferrier et al., 2000), select planning units for achieving pre-defined conservation targets (see later) or assess their relative importance for doing so. In many conservation planning studies, planning units comprise arbitrary subdivisions of grid cells (such as quarter degree squares) but cadastral boundaries and property boundaries have also been used (Pressey and Logan, 1998). The size and configuration of planning units can have important effects on the outcome of reserve selection algorithms (Pressey and Logan, 1998).

To derive a systematic conservation plan for the CFR, planning units were based on sixteen-degree squares (SDS, approximately 3900 ha). For a better representation of existing protected areas, the exact configurations of protected areas were embedded in this grid. Thus broad-scale planning units consisted of sixteen-degree squares wrapped around boundaries of protected areas. Because the fine-scale (Agulhas Plain) study was aimed at identifying implementation opportunities and constraints associated with land tenure and incentives (Pence et al., 2003), cadastral units, which are largely coincident with farm boundaries under single ownership, were used for fine-scale conservation planning in the Agulhas Plain (Table 1).

Data sets were compiled to record the occurrence of unique biodiversity features (area of BHUs or vegetation types) per planning unit (SDS or cadastral unit). Only the currently untransformed area of each biodiversity feature was recorded.

2.3. Conservation targets

Conservation targets interpret broad, qualitative conservation goals for a planning domain into quantitative guidelines for planning decisions. In both the broad- and fine-scale data sets, the goal of this study was to identify and implement a system of conservation areas that would ensure adequate representation of the region's biodiversity (Cowling et al., 1999b; Cole et al., 2000). Targets were set for each of the nine BHUs and 36 vegetation types in the study area. Targets for each feature should ideally vary according to their different needs for conservation (Pressey and Taffs, 2001). Conservation targets were allocated with the formula:

$$\mathrm{TARGET} = B + R$$

where B is a baseline target, which accommodates differential patterns of plant species turnover and R is a retention target to retain a proportion of the untransformed habitat in relation to future biodiversity threats. In the fine-scale Agulhas data set, baseline targets were 10% of original (pre-transformation) area for vlei and forest vegetation types, 15% for lowlands vegetation types and 25% for montane vegetation types. In the broad-scale Cape data sets, baseline targets were 10% of original area for lowland BHUs, and 15% for upland BHUs. Retention targets were allocated as percentages of untransformed areas as follows:

$$R = H \times (1 + [t - e]/e)$$

where H is a threat weighting component, t the original area of the biodiversity feature, e the area currently untransformed. Retention targets were therefore larger, as percentages of untransformed areas, for vegetation types or BHUs that were more threatened and had been more extensively transformed. Spatially-explicit predictions of future urbanisation, agriculture and invasive alien plants were derived (see Cole et al., 2000; Rouget et al., 2003) to assess the potential of future habitat transformation for each biodiversity feature. If the highest potential across these three threats was high, then H was 30% of untransformed habitat, if the highest potential was medium, then H was 15%, and if the highest potential was low, then H was 0% of the untransformed habitat (see Pressey et al., 2003, for details). Final conservation targets ranged from 11.7 to

100% of the untransformed area of vegetation types, and from 55 to 100% of the untransformed area of BHUs.

2.4. Spatial analysis of conservation value and priority

Reserve selection algorithms select sets of areas (based on planning units) to achieve nominated conservation targets. The resulting arrays of planning units have been termed "minimum sets" or "near minimum sets" and many conservation planning exercises have used minimum sets in the past (e.g. Kirkpatrick, 1983; Possingham et al., 2000). However, a minimum set indicates nothing about the potential contribution of unselected areas to targets or the relative importance of all the planning units (selected or unselected) in a region for achieving targets. There may be many alternative minimum sets for achieving the same targets (Pressey et al., 1997). To counter the limitations of minimum sets, a measure has been developed which reflects the relative importance of any area in contributing to the conservation target. This measure has been termed "irreplaceability" (Pressey et al., 1994; Ferrier et al., 2000). Irreplaceability values range from 0 (not needed) to 1 (irreplaceable, essential for achieving the set of conservation targets). Previous studies have shown that the choice of planning units as well as the biodiversity features influence minimum sets and irreplaceability measures (Pressey and Logan, 1998; Pressey et al., 1999).

C-Plan was used for deriving minimum sets and irreplaceability values across the Agulhas Plain using the two original and three derived data sets described earlier (see Table 1). The analysis was subdivided into two sections: (1) the extent to which conservation planning using broad-scale components (biodiversity, habitat transformation mapping, and planning units) achieve fine-scale conservation targets (identified for the 36 vegetation types); and (2) the factors driving patterns of irreplaceability at fine and broad scales.

2.4.1. Minimum sets

For each of the five data sets (Table 1), minimum sets were generated to achieve all conservation targets. The extent to which each minimum set contributed to conservation targets defined for the Agulhas Plain at the fine-scale level (Agulhas data set in Table 1) was computed. For each minimum set, the number of vegetation types for which conservation targets (as calculated in the original Agulhas data set) were achieved was calculated. An index of reserve design efficiency was derived as follows:

$$\mathrm{EFFICIENCY} = \mathrm{CONTRIB}/\mathrm{MIN\ SET}$$

where CONTRIB is the total area of selected untransformed vegetation types contributing to conservation targets for the Agulhas data set, and MIN SET is the total area of untransformed vegetation selected for each minset.

2.4.2. Irreplaceability patterns

Irreplaceability values of all planning units were generated by C-Plan for each of the five data sets (see Table 1) and compared to irreplaceability values found using fine-scale data (original Agulhas data set). Classification trees in S-Plus (Chambers and Hastie, 1992) were used to identify which variables influenced irreplaceability pattern. Classification trees are very suitable for such analysis because they can incorporate both categorical and continuous factors, and because of their ability to detect interactions and non-additive behaviour among variables (Breinam et al., 1984; Hastie et al., 2001). These non-parametric methods are also distribution-free. Previous studies have shown that classification trees generate more accurate results for analysing determinants of distribution than traditional regression techniques (De'ath and Fabricius, 2000; Rouget et al., 2001).

This study focussed more on understanding the data structure then predicting the outcomes with high accuracy; therefore classification trees were kept simple and with relatively few terminal nodes (less than 8). Trees were pruned after 6 nodes in each case.

To investigate the effect of scale of biodiversity mapping on irreplaceability pattern, two maps of irreplaceability were derived using the original Agulhas data set and the derived data set 1 (Table 1). Biodiversity features consisted of BHUs (derived data set 1) and vegetation types in the original Agulhas data set (Table 1). Fine-scale habitat transformation mapping and small planning units did not vary between data sets. Differences in irreplaceability values were analysed at the cadastral level using classification trees in S-Plus (Chambers and Hastie, 1992). All planning units less than 25 ha were ignored to account for the coarse resolution of the broad-scale data (mapped at 1:250,000). Irreplaceability values were grouped in five categories: <0.2, 0.2–0.4, 0.4–0.6, 0.6–0.8, and >0.8.

The data set consisted of 670 planning units. The dependent variable, the difference in irreplaceability values, was categorised as follows:

- 0 (match between two scales, i.e. same irreplaceability category in both data sets)
- $+1$ (over-estimate of irreplaceability from derived data set 1 using broad-scale biodiversity features)
- -1 (underestimate of irreplaceability from derive data set 1 using broad-scale biodiversity features).

The following factors were considered as potential predictors of differences in irreplaceability: the total area of the planning unit, the area of untransformed vegetation within a planning unit, the presence/absence of each BHU and vegetation type occurring in the planning unit, and the number of vegetation types present in the planning unit.

Because the planning units (cadastral units) were not homogenous in size, such unweighted analysis would give similar importance to differences in irreplaceability value for small or large planning units. Preliminary weighted analysis (weight proportional to area of planning unit) generated regression trees with similar significant factors as unweighted analysis; therefore unweighted analysis was performed throughout the study.

To investigate the effect of the scale of habitat transformation mapping on irreplaceability pattern, irreplaceability values obtained from the original Agulhas data set and derived data set 2 (Table 1) were compared. Targets for biodiversity features and the area available for conservation planning were derived from the two different estimates of habitat transformation (ground survey for the original Agulhas data and remote sensing for the derived data set). Biodiversity features and planning units were as for the original Agulhas data set-vegetation types and cadastral units, respectively (Table 1). Differences in irreplaceability values at the cadastral level were analysed using a classification tree as above. The data set consisted of 670 planning units and the same dependent and independent variables were used as earlier.

To investigate the effect of the size of planning units on irreplaceability pattern, irreplaceability values generated from the original Agulhas data set and derived data set 3 (Table 1) were compared. Planning units for the original Agulhas data set were cadastral units and those for derived data set 3 were sixteenth degree squares. Biodiversity features and mapping of habitat transformation were as for the original Agulhas data set (Table 1). Because these two data sets are based on different planning units, direct spatial comparison was not possible. Both layers of planning units were therefore intersected to obtain spatially comparable units. This resulted in 961 new polygons for which the irreplaceability values derived from the original Agulhas data set and derived data set 3 were retained. Therefore, each of the new 961 polygons had irreplaceability values derived from the cadastral unit and from the sixteenth degree square to which they belonged. Each new polygon also retained the original attributes (such as vegetation types present) of the cadastral unit and sixteenth degree square from which they originated. Differences in irreplaceability for all polygons ($n=961$) could then be related to differences in biodiversity representation within the original cadastral unit and sixteenth degree square. The analysis was done using classification tree as described earlier.

2.5. Analysis of spatial patterns of biodiversity and habitat transformation at fine and broad-scale

To investigate the effects of mapping resolution on biodiversity and habitat transformation patterns, the spatial characteristics of biodiversity features (BHUs and vegetation types) and habitat transformation mapping were also compared at the broad and fine scales. Both biodiversity layers were spatially intersected and frequency distribution of each vegetation type in relation to the nine BHUs were derived. This was then used to assess the extent to which fine-scale biodiversity features were nested within broad-scale biodiversity features—BHUs—(see variable nestedness later). The spatial match of habitat transformation pattern was also compared at both scales for the whole Agulhas Plain as well as for each vegetation type. Each layer of transformed/untransformed vegetation was converted to a 25 m grid and a confusion matrix was derived (cross-tabulation of untransformed/transformed areas) to assess the level of correspondence between the two scales (see variable mapping agreement of untransformed vegetation later).

3. Results

3.1. Efficiency of minimum sets at fine and broad-scale

The minimum set based on the original Agulhas data set, achieved all conservation targets (36 vegetation types, 67,137 ha required) in 154,014 ha (66% of the whole Agulhas Plain). This represents an efficiency of 81.6% (67,137 ha targeted in 82,293 ha of untransformed vegetation selected, Table 2). The minimum set, based on the original CAPE data set, achieved all conservation targets for the nine broad habitat units in 158,718 ha (68% of the total area). Although, these two minimum sets were derived independently, there was relatively good spatial overlap between them. The area selected for the original CAPE data set also captured a substantial amount of the area required for achieving fine-scale conservation targets. The CAPE minimum set achieved fine-scale targets for 27 vegetation types with an efficiency of 71.2% (Table 2). The effects of varying biodiversity features, habitat transformation mapping and planning units on minimum set efficiency are summarised in Table 2. The choice of biodiversity features seems to have the greatest effect on achieving fine-scale targets. The area selected by a minimum set using BHUs as biodiversity features (derived data set 1, Table 2) achieve targets for only 26 of the 36 vegetation types (missing types in Table 3) with the lowest efficiency (67.1%). The minimum set using remote-sensing mapping of habitat transformation (derived data set 2) met conservation targets for 31 vegetation types (missing types in Table 3). The minimum set based on larger

Table 2
Contribution to fine-scale conservation targets of minimum sets derived from different data sets (in Table 1)

Data sets	Factor analysed	Area required (ha)[a]	No. of veg types[b]	Contributing area (ha)[c]	Efficiency (%)[d]
Original					
Agulhas		82,293	36	67,137	81.6
CAPE		90,965	27	64,748	71.2
Derived					
Data set 1	Biodiversity	95,067	26	63,814	67.1
Data set 2	Habitat transformation	89,248	31	65,981	73.9
Data set 3	Planning units	95,114	36	67,137	70.6

[a] Untransformed area (in ha) selected in minimum set to achieve conservation targets.
[b] Number of vegetation types for which conservation targets have been achieved.
[c] Total area of untransformed vegetation contributing to fine-scale conservation targets (as defined in the Agulhas data set).
[d] Ratio of contributing area and area required (100% means that all conservation targets has been achieved in the minimum amount of land).

Table 3
Characteristics of fine-scale biodiversity features of the Agulhas Plain

	Vegetation type	Area (ha)	% Untransformed	Target (%)[a]	Nestedness[b]	Mapping agreement[c]	Failed target[d]
1	Acid Vlei (ACIDVLEI)	97.4	100.0	25.0	98.7	28.5	
2	Afromontane Forest (AFROMO)	107.8	85.7	45.9	75.8	100.0	T
3	Berzelia Riparian (BERZE)	4360.4	65.4	63.2	44.0	96.2	
4	Black Vlei (BLACKV)	146.3	81.0	30.1	34.2	92.8	
5	Dry Restioid Fynbos (DREST)	2355.8	36.3	65.8	93.1	65.7	C
6	Dune Thicket (DUNETH)	90.7	97.5	25.6	41.0	76.9	
7	Dune Fynbos (DUNFYN)	29,644.8	79.9	36.7	80.0	96.2	
8	Elim Fynbos (ELIFYN)	13,270.0	39.3	86.3	92.0	84.0	T
9	Elim Asteraceous Fynbos (ELIM)	23,256.2	15.3	100.0	81.1	78.9	C, B
10	Elim Riparian (ELIMR)	3340.9	41.7	83.4	74.1	70.4	
11	Estuary (ESTUARY)	268.2	98.9	25.2	61.9	97.4	
12	Limestone Fynbos (LIME)	10,360.3	81.8	18.3	61.9	96.6	
13	Limestone dominated by *Leucospermum muirii* (LIMMUIR)	727.3	90.2	16.6	54.7	97.1	
14	Limestone and Sand (LIMSAN)	15,918.7	59.3	67.4	38.7	96.1	C, T
15	Limestone Outcrop Fynbos (LIMSTO)	2023.8	71.5	40.2	47.5	94.9	
16	Limestone Vlei (LIMVLEI)	2.1	90.8	43.7	100.0	100.0	C, B
17	Milkwood Thicket (MILKWO)	991.7	82.6	47.3	48.7	91.0	
18	Mountain Fynbos (MTNFYN)	52,299.4	79.1	49.7	50.9	97.9	
19	Mountain Top Fynbos (MTNTOP)	7919.0	94.8	26.3	72.3	99.9	
20	Muddy Vlei (MUDDY)	1009.7	95.3	26.1	77.9	99.3	
21	Transitional Fynbos (NEUT)	4944.9	33.9	94.0	43.6	83.8	C, T, B
22	Palmiet Riparian (PALMI)	3144.0	58.1	68.3	55.6	87.9	
23	Pans (PAN)	45.0	52.7	41.0	57.2	0.0	
24	Renoster Fynbos (RENFYN)	8376.7	14.4	100.0	73.2	23.0	C, B
25	Renoster Grassland (RENGRA)	5236.7	27.0	100.0	50.4	2.2	T, B
26	Renosterveld (RENOST)	4298.7	21.4	100.0	59.1	21.2	C, B
27	Restio Bog (RESBOG)	33.0	100.0	45.0	100.0	100.0	
28	Restioid Wetland (RESTIO)	3462.8	81.8	53.7	49.4	61.0	B
29	Riparian Flood Plain (RIPFLO)	394.5	51.7	73.5	90.4	93.7	
30	Rivers (RIVER)	251.1	68.9	53.8	64.3	65.8	C
31	Salt Pan (SALTPAN)	354.5	85.2	11.7	72.7	97.8	
32	Sand Dune (SANDUN)	2460.7	99.4	25.1	88.8	99.1	
33	Sarcocornia Wetland (SARCO)	381.9	57.2	38.8	69.2	70.3	C
34	Short Reed (SHORTR)	1998.2	70.0	60.4	54.7	53.8	B
35	Tall Reed (TALLR)	3907.5	68.6	61.2	41.8	65.1	B
36	Wet Restioid Fynbos (WREST)	8517.1	59.0	67.6	63.3	84.9	

C: CAPE data set (broad-scale plan), B: derived data set no. 1 (biodiversity features), T: derived data set no. 2 (habitat transformation) (see Table 2).
[a] % of the untransformed habitat of each vegetation type required.
[b] % of each vegetation type occurring in any one BHU.
[c] Spatial agreement of habitat transformation mapping between fine and broad scale.
[d] Refers to the data set for which a minimum set did not achieve the target.

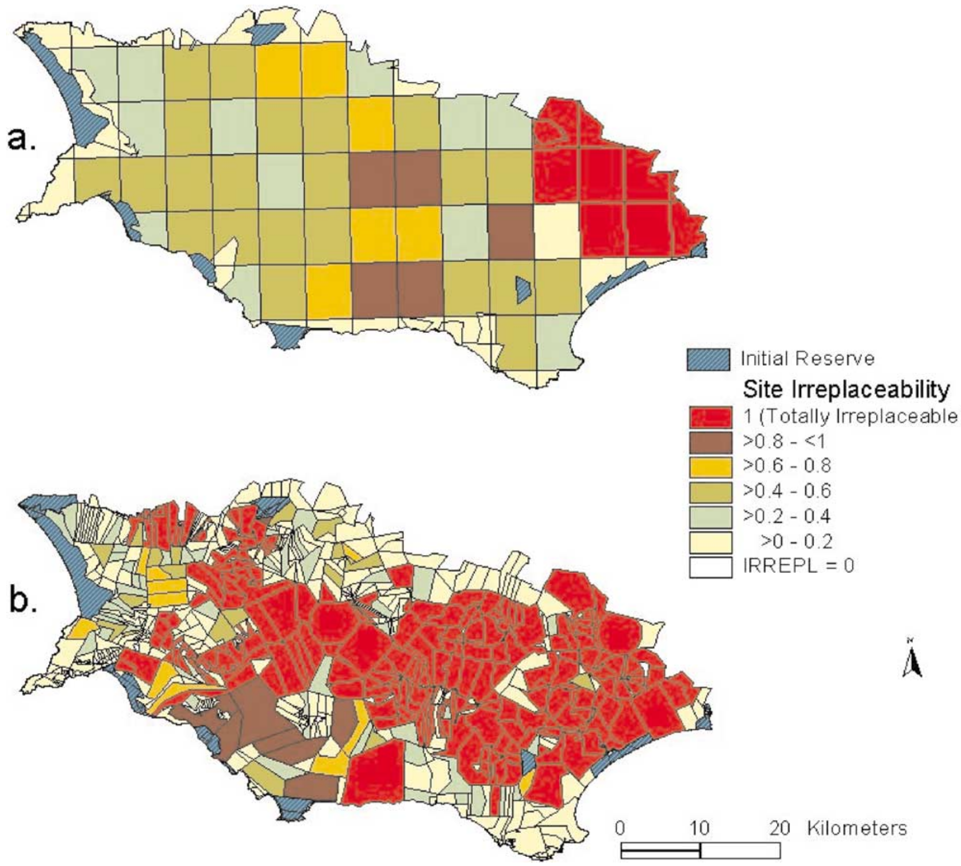

Fig. 2. Spatial pattern of irreplaceability value from: (a) broad-scale (1:250,000 scale) and, (b) fine-scale (1:10,000) conservation planning. Broad-scale data consisted of nine broad habitat units (BHUs), habitat transformation mapped by remote-sensing and planning units comprising 1/16th degree squares. Fine-scale data consisted of 36 vegetation types, habitat transformation mapped by ground survey, and cadastral boundaries as planning units.

planning units (sixteenth degree square, derived data set 3) was less efficient in achieving targets for all vegetation types compared to the minimum set for the original Agulhas data set.

3.2. What drives patterns of irreplaceability at fine and broad-scale?

Although fine-scale conservation targets could be achieved to a certain extent by minimum sets using any broad-scale component (biodiversity features, habitat transformation, or planning units), the spatial pattern of irreplaceability (the conservation value of each planning unit) differed considerably between data sets. Fig. 2 shows the differences in irreplaceability patterns obtained from the original CAPE data set and from the original Agulhas data set. Using fine-scale data, almost 50% of the total area was considered as irreplaceable (i.e. absolutely necessary to achieve conservation targets) compared to 11% when using broad-scale data. Only 18.3% of the area had similar irreplaceability value (i.e. belonging to the same irreplaceability category) (overall comparison, Table 4).

Table 4
Differences in irreplaceability values in relation to the spatial scale of biodiversity features, habitat transformation mapping and size of planning units

Comparison	Data sets	Similar values	Broad-scale overestimation	Broad-scale underestimation
Overall	Cape vs. Agulhas	18.3	27.7	54.0
Biodiversity	1 vs. Agulhas	36.3	2.9	60.8
Habitat transformation	2 vs. Agulhas	79.1	12.2	5.8
Planning units	3 vs. Agulhas	56.1	33.6	10.3

Irreplaceability values obtained from Agulhas data (fine scale) were compared to broad-scale data for the factor of concern (see Table 1). Values are percentages of the study area.

Irreplaceability values of 54% of the original Agulhas data set were underestimated with broad-scale data and 27.7% of values were overestimated (overall comparison, Table 4).

A classification tree was used to understand which factors explained the differences between irreplaceability values at fine and broad scales. Almost 80% of these differences could be explained by the presence of one vegetation type, the number of vegetation types per planning unit, and the areas of four BHUs per planning unit (Fig. 3). Differences in irreplaceability values were largely influenced by the presence of the vegetation type Elim Asteraceous Fynbos (first factor in the classification tree model, Fig. 3). Irreplaceability values derived from fine-scale data were higher where this vegetation type occurred. Elim Asteraceous Fynbos had a fine-scale conservation target of 100% of its untransformed habitat. Therefore, every planning unit where it occurred was required to achieve its target. At a broad-scale, this vegetation type was predominantly mapped as BHU Elim Fynbos/Renosterveld Mosaic, which has a lower conservation target. Irreplaceability values derived from fine-scale data were also higher in planning units where more than four vegetation types occurred. There was moreover a significant positive relationship between irreplaceability value and number of vegetation types per cadastral unit ($R^2=0.31$, $P<0.001$). Heterogeneous planning units (i.e. with high number of vegetation types) tended to contain highly transformed vegetation types—with high conservation targets—and therefore have higher irreplaceability values than more homogeneous planning units. Irreplaceability values derived from broad-scale data were higher in areas where the BHUs Elim Fynbos/Renosterveld Mosaic, Springfield Sand Plain Fynbos, and Agulhas Fynbos/Thicket Mosaic (with large targets relative to untransformed areas) occurred over relatively large areas (Fig. 3). At a fine scale, planning units where these three BHUs predominate scored lower irreplaceability values partly because of the occurrence of patches of vegetation types with low targets relative to untransformed areas.

In the following sections, more details on the effects of biodiversity features, habitat transformation mapping and planning units on irreplaceability patterns are presented.

3.2.1. BHUs vs. vegetation types

There were large differences in irreplaceability pattern between data sets using fine-scale and broad-scale biodiversity features (biodiversity comparison, Table 4). Although similar irreplaceability values were found in 36.3% of the Agulhas Plain, the use of BHUs (broad-scale biodiversity features) underestimated irreplaceability values in 60.8% of the area. In less than 3% of the area, the use of vegetation types underestimated irreplaceability value.

A classification tree using five variables could accurately explain 85% of the differences in irreplaceability value between the two data sets (Fig. 4). The presence of three vegetation types, the area of untransformed vegetation and the number of vegetation types present in each cadastral unit were the most important factors (Fig. 4). Three considerably transformed vegetation

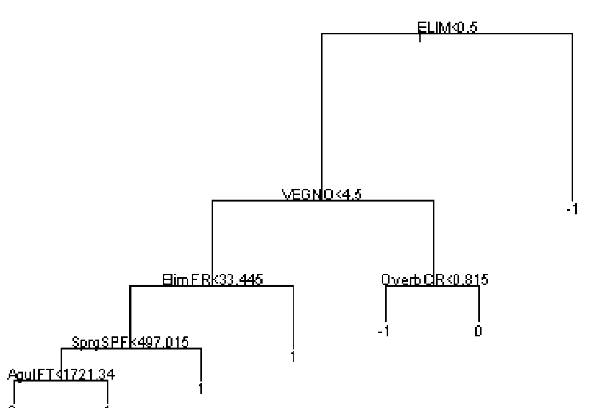

Fig. 3. Differences in irreplaceability value between broad scale and fine scale. The classification tree was generated in S-Plus. The model is based on 961 planning units (see Section 2). Areas of the vegetation type Elim asteraceous Fynbos (ELIM), and the BHUs Elim Fynbos Renosterveld Mosaic (ElimFR), Overberg Coast Renosterveld (OverbCR), Springfield Sand Plain Fynbos (SprgSPF), and Agulhas Fynbos Thicket Mosaic (AgulFT) are indicated in ha. VEGNO: number of vegetation types present in broad-scale planning units. The condition on top of the branch applies for the left side of the branch. Outcomes are predicted as follows: 0 (match between two scales, i.e. same irreplaceability category in both data sets), +1 (over-estimate of irreplaceability value from the broad-scale data set), −1 (under-estimate of irreplaceability value from the broad-scale data set).

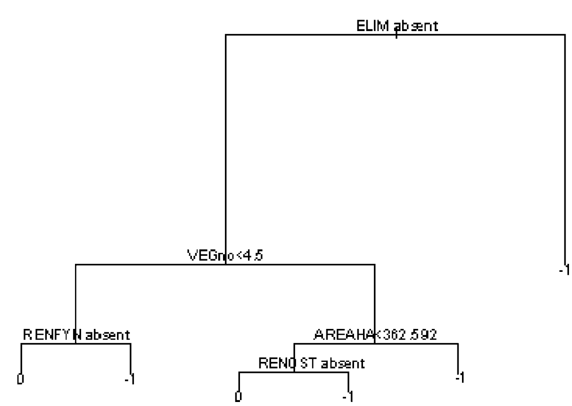

Fig. 4. Differences in irreplaceability value between broad- and fine-scale biodiversity features. The model is based on 670 cadastral units. The presence of the vegetation types Elim Asteraceous Fynbos (ELIM) Renoster Fynbos (RENFYN), and Renosterveld (RENOST) within the planning unit is indicated. AREAHA: area of the planning unit (in ha); VEGno: number of vegetation types present. The condition on top of the branch applies for the left side of the branch. Outcomes are predicted as follows: 0 (match between two scales, i.e. same irreplaceability category in both data sets), +1 (over-estimate of irreplaceability value from broad-scale data), −1 (underestimate of irreplaceability value from broad-scale data).

types that were only identified with fine-scale mapping (namely Elim Asteraceous Fynbos, Renosterveld, and Renoster Fynbos) contributed to high irreplaceability values for cadastral units in which they were present. At a broad scale, these vegetation types were mapped as widespread and common BHUs with lower conservation targets relative to untransformed areas, leading to lower irreplaceability values. Irreplaceability values were also underestimated using broad-scale biodiversity data for large cadastral units containing more than four vegetation types (Fig. 4).

3.2.2. Habitat transformation from remote sensing vs. ground survey

Irreplaceability values were relatively unaffected by fine or broad-scale mapping of habitat transformation (habitat transformation comparison, Table 4). Similar values of irreplaceability were found in almost 80% of the total area, while the use of remote sensing mapping overestimated irreplaceability in 12% of the area (Table 4). Broad-scale mapping of habitat transformation slightly increased the amount of land needed to achieve all conservation targets (Table 2). Using the classification tree model, the remaining difference in irreplaceability values within planning units could not be explained by any factor. The scale of habitat transformation mapping, however, did influence conservation targets. The use of broad-scale mapping generated higher targets for relatively untransformed vegetation types (compared to the original fine-scale targets in the Agulhas data set) and lower targets for more transformed ones (Fig. 5).

3.2.3. Sixteenth degree square vs. cadastral units

Broad-scale planning units were less efficient (i.e. more land was required) than fine-scale units for achieving all conservation targets for vegetation types (76.8% of the total area required compared to 66%, Table 2). Irreplaceability values were similar in 56% of the area, but the use of sixteenth degree squares generated higher irreplaceability values for 33.6% of the area (comparison of planning units, Table 4).

Almost 75% of the differences in irreplaceability value between derived data set 3 (using sixteenth degree squares planning units) and the original Agulhas data set (using cadastral boundaries) could be explained by a classification tree (Fig. 6). The model was based on four vegetation types and the area of untransformed vegetation in cadastral units. Irreplaceability value was mostly driven by the occurrence of one vegetation type, Elim Asteraceous Fynbos. In cases where this type was present in both planning units (cadastral units and sixteenth degree squares), the irreplaceability value was similar, irrespective of other vegetation types present (Fig. 6). The same applied for the vegetation type Renoster Fynbos. Conservation targets for these two vegetation types require all the untransformed area, therefore each planning unit where they occur became totally irreplaceable. In most cases where the vegetation types Mountain Top Fynbos and Wet Restioid Fynbos occur in sixteenth degree squares planning units but not in cadastral units, their occurrence led to higher irreplaceability values in the broad-scale planning units (Fig. 6).

3.3. Analysis of similarities and differences of basic features at broad and fine mapping scales

The two previous sections have shown the importance of the choice of biodiversity features in both determining pattern of irreplaceability and achieving conservation targets. The next section explores more generally

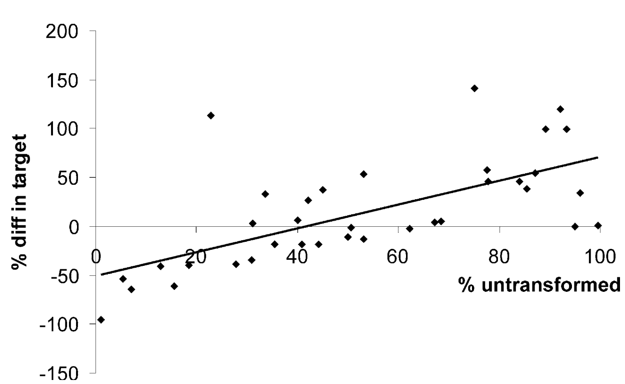

Fig. 5. The effect of habitat transformation mapping scale in setting conservation targets for vegetation types. For each vegetation type, the difference (in %) between target derived using remote sensing mapping of habitat transformation (broad scale) and target derived using field-mapping (fine scale) were calculated.

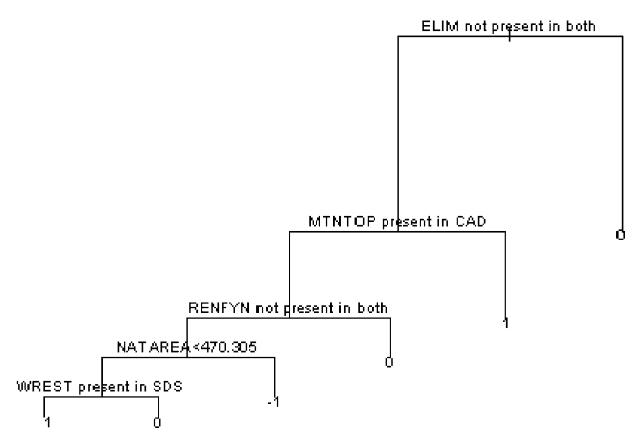

Fig. 6. Differences in irreplaceability value between broad-scale (1/16th degree square, SDS) and fine-scale (cadastral boundaries, CAD) planning units. The model is based on 961 planning units. Significant vegetation types were: Elim Asteraceous Fynbos (ELIM), Mountain Top Fynbos (MTNTOP), Renoster Fynbos (RENFYN), and Wet Restioid Fynbos (WREST). NATAREA: area of untransformed vegetation. The condition on top of the branch applies for the left side of the branch. Outcomes are predicted as follows: 0 (match between two scales, i.e. same irreplaceability category in both data sets), +1 (over-estimate of irreplaceability value from broad-scale data), −1 (underestimate of irreplaceability value from broad-scale data).

how patterns of biodiversity and habitat transformation changed from fine to broad-scale, and how these changes relate to conservation targets and irreplaceability pattern.

3.3.1. Patterns of biodiversity and habitat transformation

Conservation targets were not achieved for 15 vegetation types when minimum sets were produced from the original CAPE data set or derived data sets 1 and 2 (see Table 3). These 15 vegetation types shared similar characteristics identified by a multi-variate analysis. A very simple classification tree based on the conservation target and the untransformed percentage of each type could correctly classify 32 of the 36 vegetation types into two categories: vegetation types for which conservation targets are achieved in all cases, and vegetation types for which conservation targets are not achieved when using broad-scale components (see Fig. 7). Targets were always achieved for vegetation types with low conservation targets (target $<37.8\%$ of untransformed area) while targets for highly-transformed vegetation types with high conservation targets relative to untransformed areas could not be achieved in all cases (Fig. 7). Minimum sets based on broad-scale biodiversity features (BHUs) incidentally included untransformed portions of vegetation types and achieved conservation targets for some of them (26 out of 36, see Table 2).

3.3.2. Patterns of habitat transformation

Although the extent and configuration of habitat transformation relative to biodiversity features generated pronounced differences in efficiency and irreplaceability patterns, the spatial distribution of untransformed land available for conservation planning was not greatly

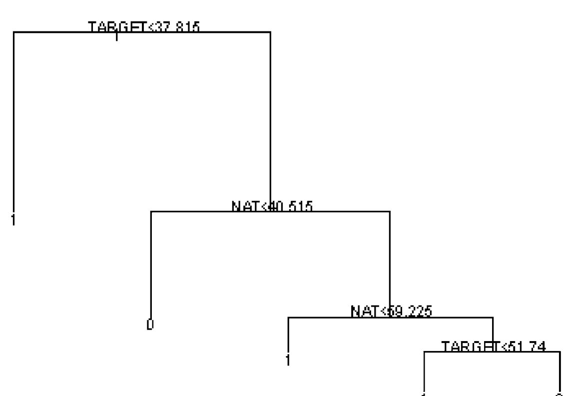

Fig. 7. Efficiency of broad-scale conservation planning in achieving conservation targets for 36 vegetation types identified at the fine scale. TARGET: original fine-scale conservation target (in% of untransformed area). NAT: untransformed area (in % of total area of vegetation type). The condition on top of the branch applies for the left side of the branch. Outcomes are predicted as follows: 0 (conservation target not achieved at broad-scale conservation planning), 1 (conservation target achieved at fine and broad-scale conservation planning).

affected by the scale of the analysis. Almost 80% of the untransformed area mapped at the fine scale (original Agulhas data set) was classified as untransformed at the broad scale (original CAPE data set). Looking at the spatial distribution of transformed areas, there was also a very good spatial match between the two data sets (86% agreement, Kappa value of 0.69).

Differences in habitat transformation mapping were not randomly distributed throughout the study area or among the 36 vegetation types. Differences were greater for the more transformed vegetation types. The variables mapping agreement and extent of untransformed vegetation were positively correlated ($r=0.52$, $P<0.01$). Habitat transformation was overestimated at the broad scale for heavily transformed vegetation types such as renosterveld, as small remnants of natural vegetation were not captured by the broad-scale study. For these vegetation types, the overestimation of habitat transformation led to a smaller area being selected at broad-scale. This resulted in fine-scale targets not being achieved in a minimum set based on the CAPE data set.

4. Discussion

This study has attempted to demonstrate some of the implications of moving from fine- to broad-scale conservation planning and vice-versa. As conservation resources are generally limited, the need for fine-scale conservation planning should be considered against a generally more rapid conservation assessment at broader scales. Careful attention should be given to whether conservation decisions will benefit from additional information gained at a finer scale (Conroy and Noon, 1996). Because of its expense per unit area, fine-scale conservation planning is usually available only over limited areas, and regional planning therefore uses broad-scale data.

Conservation value, or irreplaceability in this study, was here expressed as a function of the distribution of biodiversity features, the spatial patterns of habitat transformation, and the size of planning units. This paper attempted to tease out the effects of these interacting factors on irreplaceability value and efficiency at fine (1:10,000) and broad (1:250,000) scales. Because of the interactive nature of these factors, the results can only be indications of possible consequences of taking conservation decisions at various spatial scales and should be interpreted cautiously.

4.1. Moving from broad to fine-scale conservation planning

The analysis of selection efficiency indicated that broad-scale conservation planning was relatively effective in achieving fine-scale conservation targets (Table 2).

This raises the possibility that fine-scale data are not an absolute requirement for conservation planning. However, the vegetation types for which conservation targets were not achieved are heavily transformed, still under threat, and are therefore priorities for conservation actions. Fine-scale conservation planning is the only way to identify these threatened habitats and their associated species. Moreover, the measure of efficiency used here is relevant only if the whole plan is implemented, meaning all selected areas being reserved. In practice, implementation of conservation actions often takes place over long period (Pressey and Taffs, 2001; Pence et al., 2003). The use of minimum sets alone does not provide any information regarding the relative need of selected area for protection because they fail to identify areas as first priority for conservation in relation to irreplaceability and threat (Pressey, 1997).

As opposed to efficiency, patterns of irreplaceability (the likelihood of any planning unit being required to achieve conservation targets) revealed strong differences between fine and broad scales (Table 4). Although broad- and fine-scale conservation plans differ in many aspects (biodiversity features, habitat transformation mapping and configuration of planning units), differences in irreplaceability values could be explained by just a few factors; the same vegetation types were found to determine irreplaceability values throughout the study.

4.1.1. Biodiversity

The fine-scale conservation plan generated overall higher irreplaceability values in planning units than the broad-scale plan. About 50% of the study area was totally irreplaceable at the fine scale compared to 10% at the broad scale. Irreplaceability patterns at fine and broad scales mostly differed in areas with highly transformed vegetation types (such as Elim Asteraceous Fynbos). These habitats, occurring on small and isolated fragments, were overlooked in the broad-scale analysis. At broad scale, they were incorporated into larger biodiversity features (BHUs) with lower conservation targets, thereby underestimating irreplaceability value of these fragments. Irreplaceability value was also underestimated by the broad-scale analysis in planning units with a relatively large number of vegetation types. Overall, conservation planning at the fine-scale shifted higher irreplaceability values towards fragmented and/or diverse habitats. This suggests that the adverse effects of broad-scale conservation planning would be more severe in fragmented and/or heterogeneous habitats but reduced in relatively intact and homogeneous habitats. Fine-scale conservation planning might therefore be generally more effective in preserving habitat diversity in fragmented and/or heterogenous areas. This is supported by the findings of Stohlgren et al. (1997) who showed that sampling plant diversity in heterogeneous habitats required finer spatial resolution (i.e. minimum mapping size) than for homogeneous habitats in order to better represent biodiversity pattern. The positive relationship between habitat fragmentation and habitat heterogeneity is however likely to be specific to the Agulhas Plain, where greatest vegetation type heterogeneity, comprising remnants of Elim types (7–9 in Table 4) Renoster types (24–26) and many wetland types, is found along catenae associated with agriculturally valuable soils (Thwaites and Cowling, 1988). The resultant transformation has left small pockets of these vegetation types, some on ferricrete outcrops unsuitable for cultivation (top of the catena; Elim Asteraceous Fynbos, Elim Fynbos), others on steep high quality midslopes soils that have escaped cultivation (Renoster Fynbos, Renoster Grassland, Renosterveld), and many wetland types on bottomlands unsuitable for cultivation (R.M. Cowling, personal communication).

Although fine-scale habitat heterogeneity is partially lost in broad-scale assessment of conservation value, the latter probably generate more consistent patterns of biodiversity for regional planning. Fine-scale biodiversity mapping might not consistently record features throughout the landscape (e.g. owing to landscape heterogeneity and inaccessibility) and is generally less objective than broad-scale classification (Fuller et al., 1998). However, broad land systems tend to be more heterogeneous. Heterogeneity within and between classes has been rarely addressed for the setting of conservation targets in conservation planning (Ferrier, 2002; Pressey et al., 2003).

4.1.2. Habitat transformation

Recently, the consideration of habitat transformation in conservation planning has received increasing attention (Sisk et al., 1994; Richardson et al., 1996; Pressey, 1997; Flather et al., 1998; Rouget et al., 2003). Quantitative assessments of current and future habitat transformation improve the setting of conservation targets (Pressey et al., 2003) and the identification of conservation priorities (Pressey et al., 1996; Pressey and Taffs, 2001). In this study, habitat transformation was quantified from remote-sensing interpretation (broad scale) and ground survey (fine scale). The spatial pattern of habitat transformation was very similar at both scales (80% of mapping agreement), but there was a strong negative relationship between mapping agreement and extent of transformation for the 36 vegetation types (Table 3). The most transformed vegetation types showed large discrepancies in habitat transformation between broad-scale and fine-scale mapping.

The consideration of habitat transformation was crucial when combined with biodiversity pattern. Patterns of irreplaceability at the fine-scale were driven by the presence of few highly transformed vegetation types; the irreplaceability value of sites where these vegetation types occurred was always underestimated at the broad scale (Fig. 3). At the broad scale, underestimation of irreplaceability value in fragmented habitats will obviously depend on the degree of fragmentation (average size of fragments and connectivity between fragments). In this study, habitat transformation pattern was available at two spatial scales only. By analysing the effects of various resolutions (window size) on landscape characteristics in the Everglades region, Obeysekera and Rutchey (1997) found that islands of natural vegetation almost disappear beyond the 700 m resolution. Very few fragments of natural vegetation were identified using a window size of 700 m. The critical mapping scale for which irreplaceability values considerably drop in different regions needs further investigation.

4.1.3. Planning units

Most approaches to systematic conservation planning rely on planning units with various sizes and shapes. This study confirms that broad-scale planning units (in this case, 1/16th degree square) are less efficient in achieving conservation targets than smaller units (and see Pressey and Logan, 1998; Rodrigues and Gaston, 2001). The minimum set based on 1/16th degree squares selected more land than the minimum set based on cadastral units to fulfil the same conservation targets (Table 2). Large planning units are therefore more costly to implement (Pressey and Logan, 1998). Small units, such as cadastral units, can also be amalgamated with much more flexibility in choices of boundaries and configuration than larger units. When these small units are linked to land tenure, they greatly facilitate the implementation phase (Pence et al., 2003).

4.2. Is there an appropriate spatial scale for conservation planning?

Because of its hierarchical nature, biodiversity can be depicted at several levels of spatial organisation (Franklin, 1993; Humphries et al., 1995). Several studies on biodiversity patterns have suggested that results obtained from one spatial scale might not be applicable to another scale (Levin, 1992; Collingham et al., 2000). This leads to the following question: is there an appropriate scale for conservation planning? From this study, one could easily argue that mapping biodiversity at 1:10,000 is still too coarse, and a finer mapping resolution is required to capture most of the biodiversity pattern. A finer mapping resolution would identify very small habitat types, some of which are highly transformed. As the results of this study suggest, these highly transformed habitats of high conservation value are the most likely to be overlooked by a broad-scale conservation plan. These habitats would be considered of high priority if conservation planning were undertaken at a finer resolution.

There is, however, no clear theoretical or practical answer on which scale to measure biodiversity (Heywood, 1995; Nagendra and Gadgil, 1999) and a multiple-scale approach is probably required (Conroy and Noon, 1996; Poiani et al., 2000). Any single-scale conservation assessment is likely to be flawed because of the hierarchical nature of biodiversity operating at various spatial scales (Fairbanks and Benn, 2000). This study has shown that fine-scale biodiversity patterns are not entirely nested within broader units and the irreplaceability value of the same site differs between fine- and broad-scale conservation planning. At the broad scale, irreplaceability values were thus underestimated in 54% of the area of the Agulhas Plain and over-estimated in 27.7% of the study area.

It is generally stated that coarse-filter approaches encompass most of the biodiversity levels (Noss, 1987), but this has been tested in very few studies (Wessels et al., 1999; Araujo et al., 2001). Land classes (such as BHUs) appear to be good surrogates for biodiversity (Cowlings and Heijnis, 2001; Lombard et al., 2003) but they are only surrogates. Conservation planning based on BHUs or other surrogates should be complemented by more fine-scale and detailed biodiversity assessment. In a previous conservation study on the Agulhas Plain, Lombard et al. (1997) found that reserve selection algorithms based on 11 vegetation types adequately represented populations of most endemic species. Systematic conservation planning should therefore aim at representing biodiversity patterns in its hierarchical form (from land classes to species) and incorporating biodiversity persistence (Cowling et al., 1999a). A good conservation plan should also look at combinations of land classes and the interactions between them.

4.3. Conclusion

Outcomes of conservation planning are scale-dependent. They rely on the spatial scale at which biodiversity features and habitat transformation are mapped and at which planning units are delineated. This study suggests that broad-scale conservation planning is probably suitable for homogenous and relatively intact landscapes. Such scale of analysis provides quick and consistent assessment of conservation value for entire regions. However, conservation value is likely to be underestimated in areas where localised and heavily transformed fine-scale habitats do occur. Fine-scale conservation planning is thus most important in fragmented and heterogeneous landscapes where the limitations of broad-scale mapping are likely to be greatest.

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References

Araujo, M.B., Humphries, C.J., Densham, P.J., Lampinen, R., Hagemeijer, W.J.M., Mitchell-Jones, A.J., Gasc, J.P., 2001. Would environmental diversity be a good surrogate for species diversity? Ecography 24, 103–110.

Beckett, P.H.T., Burrough, P.A., 1971. The relation between cost and utility in soil survey. IV. Comparison of the utilities of soil maps produced by different procedures, and to different scales. Journal of Soil Science 22, 466–480.

Breinam, L., Friedman, J.H., Olshen, R.A., Stone, C.J., 1984. Classification and Regression Trees. Wadsworth, Belmont.

Burke, V.J., 2000. Landscape ecology and species conservation. Landscape Ecology 15, 1–3.

Chambers, J.M., Hastie, T.J., 1992. Statistical models in S. Pacific Grove.

Christensen, N.L., Bartuska, A.M., Brown, J.H., Carpenter, S., D'Antonio, C., Francis, R., Franklin, J.F., MacMahon, J.A., Noss, R.F., Parsons, D.J., Peterson, C.H., Turner, M.G., Woodmansee, R.G., 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications 6, 665–691.

Cole, N.S., Lombard, A.T., Cowling, R.M., Euston-Brown, D., Richardson, D.M., Heijnis, C.E., 2000. Framework for a Conservation Plan for the Agulhas Plain, Cape Floristic Region, South Africa. IPC Report 00/01. Institute for Plant Conservation, University of Cape Town.

Collingham, Y.C., Wadsworth, R.A., Huntley, B., Hulme, P.E., 2000. Predicting the spatial distribution of non-indigenous riparian weeds: issues of spatial scale and extent. Journal of Applied Ecology 37, 13–27.

Conroy, M.J., Noon, B.R., 1996. Mapping of species richness for conservation of biological diversity: conceptual and methodological issues. Ecological Applications 6, 763–773.

Cowling, R.M., 1992. The Ecology of Fynbos: Nutrients, Fire and Diversity. Oxford University Press, Cape Town.

Cowling, R.M., Campbell, B.M., Mustart, P.J., McDonald, A.P., Jarman, M.L., Moll, E.J., 1988. Vegetation classification in a floristically complex area: the Agulhas Plain. South African Journal of Botany 54, 290–300.

Cowling, R.M., Heijnis, C.E., 2001. The identification of Broad Habitat Units as biodiversity features for a systematic conservation planning in the Cape Floristic Region. South African Journal of Botany 67, 15–38.

Cowling, R.M., Holmes, P.M., 1992. Endemism and speciation in a lowland flora from the Cape Floristic Region. Biological Journal of the Linnaean Society 47, 367–383.

Cowling, R.M., Pressey, R.L., 2003. Introduction to systematic conservation planning in the Cape Floristic Region. Biological Conservation 112, 1–13.

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

Cowling, R.M., Pressey, R.L., Lombard, A.T., Heijnis, C.J., Richardson, D.M., Cole, N., 1999b. Framework for a Conservation Plan for the Cape Floristic Region. IPC Report 9902. Institute of Plant Conservation, University of Cape Town.

Davis, F.W., 1995. Information systems for conservation research, policy and planning. BioScience Supplement S36–S42.

De'ath, G., Fabricius, K.E., 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81, 3178–3192.

Fairbanks, D.H.K., Benn, G.A., 2000. Identifying regional landscapes for conservation planning: a case study from KwaZulu-Natal, South Africa. Landscape and Urban Planning 50, 237–257.

Ferrier, S., 2002. Mapping spatial pattern in biodiversity for regional conservation planning: where to from here? Systematic Biology 51, 331–363.

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

Flather, C.H., Knowles, M.S., Kendall, I.A., 1998. Threatened and endangered species geography. BioScience 48, 365–376.

Franklin, J.F., 1993. Preserving biodiversity—species ecosystems or landscapes? Ecological Applications 3, 202–205.

Fuller, R.M., Wyatt, B.K., Barr, C.J., 1998. Countryside survey from ground and space: different perspectives, complementary results. Journal of Environmental Management 54, 101–126.

Hastie, T., Friedman, J., Tibshirani, R., 2001. The Elements of Statistical Learning. Springer-Verlag, New York.

Heydenrych, B.J., Cowling, R.M., Lombard, A.T., 1999. Strategic conservation interventions in a region of high biodiversity and high vulnerability: a case study from the Agulhas Plain at the southern tip of Africa. Oryx 33, 256–269.

Heywood, V.H. (Ed.), 1995. Global Biodiversity Assessment. Cambridge University Press, Cambridge.

Humphries, C., Williams, P.H., Vane-Wright, R.I., 1995. Measuring biodiversity value for conservation. Annual review of Ecology and Systematics 26, 93–111.

Kirpatrick, J.B., 1983. An iterative method for establishing priorities for selection of nature reserves: an example from Tasmania. Biological Conservation 25, 127–134.

Lambeck, R.J., 1997. Focal species: a multi-species umbrella for nature conservation. Conservation Biology 11, 849–856.

Levin, S.A., 1992. The problem of pattern and scale in ecology. Ecology 73, 1943–1967.

Lloyd, J.W., van den Berg, E.C., van Wyk, E., 1999. CAPE Project. The Mapping of Threats to Biodiversity in the Cape Floristic Region with the Aid of Remote Sensing and Geographic Information Systems. Report GW/A1999/54. Agricultural Research Council, Institute for Soil, Climate and Water, Pretoria, South Africa.

Lombard, A.T., Cowling, R.M., Pressey, R.L., Rebelo, A.V., 2003. Efficiency of land class versus species locality data in conservation planning for the Cape Floristic Region. Biological Conservation 112, 45–62.

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

Mittermeier, R.A., Myers, N., Thomsen, J.B., 1998. Biodiversity hotspots and major tropical wilderness areas: approaches to setting conservation priorities. Conservation Biology 12, 516–520.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots conservation priorities. Nature 403, 853–858.

Nagendra, H., Gadgil, M., 1999. Biodiversity assessment at multiple scales: linking remotely sensed data with field information. Proceedings of the national Academy of the United States of America 96, 9154–9158.

Noss, R.F., 1987. From plant communities to landscape in conservation inventories: a look at The Nature Conservancy (USA). Biological Conservation 41, 11–37.

Obeysekera, J., Rutchey, K., 1997. Selection of scale for Everglades landscape models. Landscape Ecology 12, 7–18.

Pence, G., Botha, M., Turpie, J.K., 2003. Evaluating combinations of on-and off-reserve conservation strategies for the Agulhas Plain, South Africa: a financial perspective. Biological Conservation 112, 253–273.

Poiani, K.A., Richter, B.D., Anderson, M.G., Richter, H.E., 2000. Biodiversity conservation at multiple scales: functional sites, landscapes, and networks. BioScience 50, 133–146.

Possingham, H., Ball, I., Andelman, S., 2000. Mathematical methods for identifying representative reserve networks. In: Ferson, S., Burgman, M. (Eds.), Quantitative Methods for Conservation Biology. Springer-Verlag, New York, pp. 291–305.

Pressey, R.L., Taffs, K.H., 2001. Scheduling conservation action in production landscapes: priority areas in western New South Wales defined by irreplaceability and vulnerability to vegetation loss. Biological Conservation 100, 355–376.

Pressey, R.L., 1997. Priority conservation areas: towards an operational definition for regional assessments. In: Pigram, J.J., Sundell, R.C. (Eds.), National Parks and Protected Areas: Selection, Delimitation and Management. Center for Water Research Policy, University of New England, Armidale, pp. 337–357.

Pressey, R.L., Bedward, M., 1991. Mapping the environment at different scales: benefits and costs for nature conservation. In: Margules, C.R., Austin, M.P. (Eds.), Nature Conservation: Cost Effective Biological Surveys and Data Analysis. CSIRO, Melbourne, pp. 7–13.

Pressey, R.L., Cowling, R.L., Rouget, M., 2003. Formulation of conservation targets for biodiversity pattern and process in the Cape Floristic Region. Biological Conservation 112, 99–127.

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

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

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

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

Pressey, R.L., Possingham, H.P., Logan, V.S., Day, J.R., Williams, P.H., 1999. Effects of data characteristics on the results of reserve selection algorithms. Journal of Biogeography 26, 179–191.

Richardson, D.M., van Wilgen, B.W., Higgins, S.I., Trinder-Smith, T.H., Cowling, R.M., McKell, D.H., 1996. Current and future threats to plant biodiversity on the Cape Peninsula, South Africa. Biodiversity and Conservation 5, 607–647.

Rodrigues, A.S.L., Gaston, K.J., 2001. How large do reserve networks need to be? Ecology Letters 4, 602–609.

Rouget, M., Richardson, D.M., Cowling, R.M., Lloyd, J.W., Lombard, A.T., 2003. Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape Floristic Region, South Africa. Biological Conservation 112, 63–85.

Rouget, M., Richardson, D.M., Milton, S.J.M., Polakow, D., 2001. Predicting the dynamics of four invasive *Pinus* species in a fragmented semi-arid shrubland in South Africa. Plant Ecology 152, 79–92.

Rouget, M., Richardson, D.M. Understanding actual and potential patterns of plant invasion at different spatial scales: quantifying the roles of environment and propagule pressure. In: Child, L. et al. (Eds.), Proceedings of the 6th International Conference on Invasive Alien Plants. Backhuys Publishers, Leiden (in press).

Rowe, J.S., Sheard, J.W., 1981. Ecological land classification: a survey approach. Environmental Management 5, 451–464.

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

Sisk, T.D., Launer, A.E., Switky, K.R., Ehrlich, P.R., 1994. Identifying extinction threats: global analyses of the distribution of biodiversity and the expansion of the human entreprise. BioScience 44, 592–604.

Soulé, M.E., Terborgh, J., 1999. Conserving nature at regional and continental scales- a scientific program for North America. BioScience 49, 809–817.

Stohlgren, T.J., Chong, G.W., Kalkhan, M.A., Schell, L.D., 1997. Multiscale sampling of plant diversity: effects of minimum mapping unit size. Ecological Applications 7, 1064–1074.

Thwaites, R.N., Cowling, R.M., 1988. Landscape vegetation relationships on the Agulhas Plain. Catena 15, 333–346.

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

Younge, A., Fowkes, S., 2003. The Cape Action Plan for the Environment: overview of an ecoregional planning process. Biological Conservation 112, 15–28.