The effect of spatial scale on the use of biodiversity surrogates and socio-economic criteria in systematic conservation assessments

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Declaration

I, the undersigned, hereby declare that the work contained in this thesis is my own original work and that I have not previously in its entirety or in part submitted it at any university for a degree.

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ABSTRACT

A systematic conservation assessment is the first phase of a systematic conservation planning protocol; it uses spatial data and representation targets for the setting of priority areas and the assessment of risk to biodiversity. This thesis describes the findings of investigations on the use of data in systematic conservation assessments.

Conservation planning can be done at different spatial scales (from global to local). Systematic Conservation planning can be done at different spatial scales (from global to local). Systematic conservation assessments rely on the use of surrogates for biodiversity and often, as well, socio-economic criteria. Biodiversity surrogates can be classified as taxonomic, community and environmental. In Chapter 2, a literature review was performed (i) to quantify the use of biodiversity surrogates and socio-economic criteria in conservation assessments; and (ii) to test the hypothesis that surrogates are chosen in respect to the hierarchical organisation of biodiversity. In other words, fine scale conservation assessments are correlated with taxonomic surrogates, large scale conservation assessments are correlated with environmental surrogates, and assemblage surrogates are assessed at an intermediary scale. The literature review was based on a structured survey of 100 ISI journal publications. The analysis revealed that spatial scale had a weak effect on the use of biodiversity surrogates in conservation assessments. Taxonomic surrogates were the most used biodiversity surrogates at all scales. Socioeconomic criteria were used in many conservation assessments. I argue that it is crucial that assemblage and environmental data be more used at larger spatial scales.

The allocation of conservation resources needs to be optimised because resources are scarce. A conservation assessment can be a lengthy and expensive process, especially when conducted at fine-scale. Therefore the need to undertake a fine-scale conservation assessment, as opposed to a more rapid and less expensive broader one, should be carefully considered. The study of Chapter 3 assessed the complementarity between regional- and local-scale assessments and the implications on the choice of biodiversity features at both scales. The study was undertaken in Réunion Island. A biodiversity assessment was performed at a regional scale and measured against a finer-scale assessment performed over a smaller planning domain. Two datasets composed of species distributions, habitat patterns and spatial components of ecological and evolutionary processes were compiled as biodiversity surrogates at each scale. Targets for local-scale processes were never met in regional assessments, while threatened species and fragmented habitats were also usually missed. The regional assessment targeting habitats represented a high proportion of local-scale species and habitats at target level (67%). On the contrary, the one targeting species was the least effective. The results highlighted that all three types of surrogates are necessary. They further suggested (i) that a spatial strategy based on a complementary set of coarse filters for regional-scale assessments and fine filters for local-scale ones can be an effective approach to systematic conservation assessments; and (ii) that information on habitat transformation should help identify where efforts should be focused for the fine-scale mapping of fine filters.

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Together with priority-area setting, the identification of threatened biodiversity features has helped to prioritise conservation resources. In recent years, this type of assessment has been applied more widely at ecosystem-level. Ecosystems can be categorised into critically endangered, endangered and vulnerable, following the terminology of the IUCN Red List of threatened species. Various criteria such as extent and rate of habitat loss, species diversity and habitat fragmentation can be used to identify threatened ecosystems. An approach based only on the criterion of the quantification of habitat loss was investigated in Chapter 4 for the Little Karoo, South Africa. Habitat loss within ecosystem type is quantified on land cover information. The study analysed the sensitivity of the categorisation process to ecosystem and land cover mapping, using different datasets of each. Three ecosystem classifications and three land cover maps, of different spatial resolutions, were used to produce nine assessments. The results of these assessments were inconsistent. The quantification of habitat loss varied across land cover databases due to differences in their mapping accuracy. It was reflected on the identification of threatened ecosystems of all three ecosystem classifications. Less than 14% of extant areas were classified threatened with the coarsest land cover maps, in comparison to 30% with the finest one; and less than 9% of ecosystem types were threatened with the coarsest land cover maps, but between 15 and 23% were threatened with the finest one. Furthermore, the results suggested that the identification of threatened ecosystems is more sensitive to the accuracy of habitat loss quantification than the resolution of the ecosystem classification. Detailed land cover mapping should be prioritised over detailed ecosystem maps for this exercise.

This thesis highlighted the importance of ecosystems and processes as biodiversity surrogates in conservation assessments and suggested that results of conservation assessments based on these data, should be more widely presented in published articles. Finally, it also made apparent the important role of mapping habitat transformation for systematic conservation plans.

ABSTRAK

Hierdie tesis beskryf die bevindinge van ondersoeke aangaande die gebruik van sistematiese grondbewaring evaluasie. In hoofstuk 2 is ‘n literatuur oorsig uitgevoer om 1) die aantal gebruiksgevalle van biodiversiteit plaasvervangers en sosio-ekonomiese faktore in grondbewaring evaluasies aan te dui; 2) om die effek van ruimtelike skaal te bepaal en die veronderstelling te toets dat plaasvervangers gekies word op grond van die rangorde van biodiversiteit. ‘n Honderd ISI joernaal publikasies is nageslaan. Die ondersoek het ‘n swak verhouding van ruimtelike skaal op die gebruik van biodiversiteit plaasvervangers in grondbewaring evaluasie getoon. Teen alle skale is taksonomiese plaasvervangers die meeste gebruik. Sosio-ekonomiese faktore is in vele gevalle toegepas. Hierdie ondersoek redeneer dat dit krities is om versamelings- en omgewingsdata vir groter ruimtelike skale te gebruik.

Klein-skaal grondbewaring evaluasie kan, in vergelyking met groter-skaal evaluasie, langer

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neem en meer kos. Hoofstuk 3 bevraagteken die beste benutting van grondbewaring hulpbronne, deur die ooreenkomste tussen streek – en plaaslike skaal evaluasies, en die implikasies op die keuse(s) van biodiversiteit plaasvervangers teen beide skaal. Hierdie ondersoek is uitgevoer in Réunion Island. Streek-skaal evalusie is gebaseer op habitat, spesies en/of ontwikkeling. Die voorkoms van dieselfde tipe biodiversiteit plaasvervangers teen plaaslike skaal word geskat. Resultate het getoon 1) dat al drie tipes plaasvervangers noodsaaklik is; 2) dat ‘n ruimtelike strategie gebaseer op growwe filters vir streek-skaal evaluering en fyn filters vir plaasike-skaal evaluering ‘n effektiewe oplossing bied vir sistematiese grondbewaring evalusie; en 3) dat inligting aangaande habitatsveranderinge moontlik nuttig is vir klein-skaal kartering van fyn filters.

Gesamentlik met sistematiese grondbewaring evaluasies, het die identifikasie van bedreigde kenmerke gehelp om meer klem te plaas op grondbewaring hulpbronne. Verskeie faktore soos omvang en tempo van habitatsverlies, spesie verskeidenheid en habitatsverbryseling kan toegepas word om bedreigde ekosisteme te identifiseer. In hoofstuk vier word die evaluasie senstiwiteit teenoor ekosisteme en landbedekkingskartering ondersoek, uitsluitlik gebaseer of die tempo van habitatsverlies. Drie ekosisteem klassifikasies en drie landbedekkingskaarte is getoets, in die Klein Karoo (Suid Afrika). Habitatsverlies is oneweredig aangedui oor landbedekkingskaarte - 30% van huidig bestaande areas is as bedreig geklassifiseer met die fynste landbedekkingskaart, en minder as 14% met die grofste kaarte. Die resultate het getoon dat 1) die identifikasie van bedreigde ekosisteme is meer sensitief aan die akuraatheid van landbedekkingskartering as aan die resolusie van die ekosisteem klassifikasie; en dat 2) omstandige landbedekkingskartering meer belangrik behoort te wees as omsandige ekosisteemskaarte vir hierdie ondersoek.

Hierdie tesis bring na vore die belangrike rol van ekosisteem kartering, prosesse en habitatsveranderinge vir sistematiese grondbewaring beplanning en is van mening dat beter pogings gemaak moet word om resultate van sulke ondersoeke te kommunikeer.

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Chapter 1

Introduction

1. Background and rationale

The research presented in this thesis was undertaken in the field of conservation planning, a sub-discipline of conservation science, that deals with the location and design of protected areas for in situ conservation of biodiversity and the implementation of conservation action on ground.

Every place on Earth is worth protecting for biodiversity conservation (Sarkar and Margules, 2002). But, ongoing exploitation needs from human societies impose forms of land use that are far more economically competitive than conservation (Margules and Pressey, 2000). Hence, places have to be selected strategically in order to represent a maximum of biodiversity.

Ad hoc reservation does not achieve this goal because it is biased towards the protection of particular subsets of biodiversity and is driven by anthropocentric motivations (Pressey, 1994). To alleviate these faults, systematic approaches to conservation planning have been developed (Pressey et al., 1993). Systematic conservation planning aims at preserving a viable sample of all biodiversity, while taking into account the fact that conservation resources are scarce and need to be used efficiently (Margules and Pressey, 2000). Areas are assessed on the basis of explicit measures of their conservation value (Pressey et al., 1993). The strengths of systematic approaches are that they are data- and target-driven, repeatable and defendable (von Hase et al., 2003).

A systematic conservation planning protocol is organised in two major phases, viz. the conservation assessment and the implementation (Knight et al., 2006a). In brief, a conservation assessment is the technical exercise of compiling and analysing biological and socio-economic spatial data in order to assess the conservation value of areas in the planning domain and to prioritise them for their conservation (Margules and Pressey, 2000; Knight et al., 2006b). It is undertaken at various spatial scales (Driver et al., 2003; Sarkar et al., 2006) and generates an array of products such maps of, for instance, priority areas and threatened ecosystems, and guidelines (Driver et al., 2005; Pierce et al., 2005).

Research in conservation planning has mostly been focused on the assessment phase

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(Knight et al., 2006a). This scientific effort has led to the development of techniques that provide a sound conceptual and technical back-up to the undertaking of this exercise. This resulted in explicit protocols (e.g. Groves et al., 2002; Cowling et al., 2003), key principles (e.g. complementarity, efficiency; see Pressey et al., 1993; Gaston et al., 2002; Margules et al., 2002) and selection algorithms implemented in powerful software packages (e.g. C-Plan, Marxan) to assist and facilitate decision support (Pressey and Cowling, 2001; Sarkar et al., 2006). However, shortcomings in the availability and reliability of input data (Noss, 2002; Sarkar et al., 2006) cast doubt on the robustness of assessment outputs at effectively depicting the conservation value of biodiversity.

Overall, the inappropriate quantity and quality of data used in conservation assessments (Ferrier, 2002) is explained by a suite of socioeconomic, technical and scientific filters (Fig. 1). Low budgetary and human resources (Mace et al., 2006), coupled with limited time for action (Meffe, 2001), impede the generation of scientific knowledge on biodiversity, leading to data that are biased taxonomically and spatially (Crane and Bateson, 2003). In addition, owing to the fact that conservation planning is a spatial exercise (Margules and Pressey, 2000), data are only practical to conservation planners if supplied in the form of geographic information systems (GIS), have the adequate spatial resolution for the scale of the assessment, and are spatially consistent across the region of interest (Noss, 2002; Ferrier, 2002).

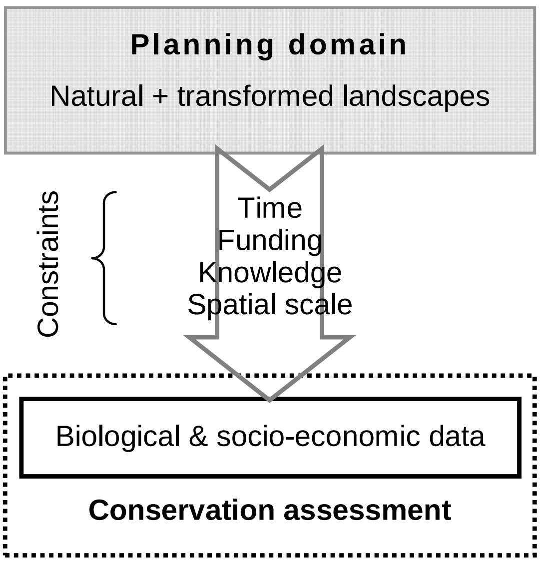

Fig. 1 – The compilation of biological and socio-economic data in conservation assessments is impeded by a series of contextual constraints.

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Of particular concern, is the limited biodiversity data which act as surrogate for overall biodiversity (Sarkar and Margules, 2002). Biodiversity surrogates should be chosen in order to represent all the levels of the biodiversity hierarchy (Noss, 1990), but, in practicality, the choice of biodiversity surrogates has been a controversial issue (see Franklin, 1993; and Brooks et al., 2004a, b; Cowling et al., 2004; Higgins et al., 2004; Molnar et al., 2004; Pressey, 2004). Research that tests surrogacy properties does not carry a universal and clear message on what are the best surrogates to use to represent a maximum of biological diversity in conservation networks (Reyers and van Jaarsveld, 2000).

More to the point, the choice of data to use as biodiversity surrogates and socio-economic criteria in conservation assessments, is often capped by budgetary and time constraints. Comparing datasets and identifying which ones provide a gain in the robustness of assessment, is a crucial point in systematic conservation planning (Stoms et al., 2005). Hence, what data to choose in conservation assessments is a relative, rather than absolute question. Insights on this question should undoubtedly be valuable to any conservation assessment exercise.

2. Objectives

This thesis looks at some of the implications of the choice of biodiversity surrogates and socio-economic criteria in conservation assessments. Conservation assessments were performed to identify priority areas and threatened ecosystems. The choice of biodiversity surrogates and socio-economic criteria is investigated through four chapters, consisting of a literature review, two analytical studies and a concluding chapter.

1.1. Literature review

Conservation assessments are undertaken at various spatial scales in order to assess the conservation value of landscapes in different geographic configurations (Erasmus et al., 1999; Driver et al., 2003). In the literature review, the relationship between spatial scale and the use of input data in conservation assessments is analysed.

My objective was to quantify the use of biodiversity surrogates and socio-economic criteria in conservation assessments and to find out how the spatial scale of the planning domain affected the choice of these data. The hypothesis tested here was that biodiversity surrogates are chosen in respect of the hierarchical organisation of biodiversity, i.e. that the larger the planning domain, the more environmental data are used and the less taxonomic data

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are used, and vice versa. This is supported, firstly, by the hierarchical organisation of biodiversity (Noss, 1990), and, secondly, by the fact that the acquisition of datasets in a conservation assessment is affected by its spatial scale and that environmental data are often the most convenient data to be generated consistently over extended areas (Ferrier, 2002).

I argue that this hypothesis provides an indirect estimation of how adequate conservation assessments are, at effectively representing all biodiversity. I tested the hypothesis in a literature review of ISI journal publications. The surveying of the articles was systematic, based on a predefined grid of parameters that described the spatial scale of the planning domain and the datasets used in all conservation assessments.

1.2. Regional- vs. local-scale conservation assessments in Réunion Island

Chapter three also explicitly looks at the relationship between spatial scale and the choice of input data (here, biodiversity surrogates only) for priority-area setting. My objective was to consider the complementarity between regional- and local-scale assessments and test where refining the spatial scale of the assessment might present a gain in the representation of biodiversity in a conservation network. This was pursued as a surrogacy analysis (i.e. testing how one biodiversity surrogate stands for another biodiversity surrogate). The methodology applied for the conservation assessments followed systematic conservation planning principles.

I use Réunion Island (France, Indian Ocean) as a case study. Two sets of distribution maps were used of biodiversity surrogates for biodiversity patterns and ecological and evolutionary processes: one set was mapped at regional scale (i.e. the largest extent) and the other at local scale. On one hand, I investigated how the priority areas identified at regional-scale incidentally represented the local-scale biodiversity surrogates. On the other hand, I tested how the patterns of irreplaceability values obtained for the regional- and local-scale surrogates were correlated. The findings were discussed in terms of complementarity of the assessments at both scales, highlighting what local-scale surrogates are a requisite in the local-scale assessment, providing that a regional-scale assessment already exists.

1.3. Land cover vs. ecosystem mapping in the assessment of threatened ecosystems in the Little Karoo

The objective of Chapter four, was to test the effect of data quality in the assessment of the conservation status of ecosystems (conservation status is employed here similarly to the

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IUCN Red List). The methodology applied was data- and target-driven, to follow some systematic conservation planning principles.

I conducted a case study on the Little Karoo region in South Africa. Three ecosystem maps, and three land cover maps of different spatial resolutions and accuracy were used. Each map was derived on different classification systems and obtained from different sources. I investigated how the different combinations of ecosystem vs. land cover maps affected the identification of the conservation status of ecosystems. The distribution of threatened ecosystems in the study area was also analysed. The results were discussed in terms of the trade-off between the use of an accurate land cover map and a fine classification of ecosystems to generate accurate ecosystem status information.

3. Thesis structure

This thesis has been written as independent manuscripts. I intend to submit Chapter 2 and 4 to *Biological Conservation,* and Chapter 3 to *Journal of Conservation Planning*, after further editing. Therefore, each chapter has its own introduction, methods, results and discussion section. This explains possible repetition between chapters.

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Chapter 2

Review

Choice of appropriate surrogates in terrestrial conservation assessments: does spatial scale matter?

A B S T R A C T

Conservation planning can be done at different spatial scales (from global to local). A systematic conservation assessment relies on the use of surrogates for biodiversity and often, as well, socio-economic criteria. Three classes of biodiversity surrogates were here identified: taxonomic, assemblage and environmental surrogates. This literature review was performed (i) to quantify the use of biodiversity surrogates and socio-economic criteria in conservation assessments; (ii) to test the hypothesis that surrogates are chosen in relation to the hierarchical organisation of biodiversity, i.e. that fine scale conservation assessments are coarsely correlated with taxonomic surrogates, large scale conservation assessments are correlated with environmental surrogates, and that assemblage surrogates are assessed at an intermediary scale. The literature review was based on a structured survey of 100 ISI journal publications. A range of information on the spatial scale of the planning domain and the nature of the input data was compiled and analysed. Spatial scale had a weak effect on the use of biodiversity surrogates: the choice of surrogates was very similar across scales, except, to some extent, at a fine scale. Assessments at intermediate and large scales were based almost exclusively on species data, most frequently of vertebrates. Fine-scale assessments used less taxonomic surrogates and more assemblage and environmental surrogates and socio-economic criteria. I argue that if conservation planning is indeed practised for the representation and persistence of all biodiversity, it is crucial that more assemblage and environmental data be used at large spatial scales.

Keywords: Biodiversity hierarchy; Literature review; Spatial extent; Spatial resolution

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1. Introduction

The protection and maintenance of biodiversity relies largely on in situ conservation (Soulé, 1991), the entire planet cannot be protected from human impacts (Sarkar and Margules, 2002). In situ conservation consists of conserving networks of sites managed through a range of on- and off-reserve strategies according to the level of security required for each site (Pence et al., 2003). The effective representation of all biodiversity requires that the process of locating priority areas be guided by the distribution of natural features (Pressey, 1994). In practice, this is performed on the basis of partial measures used as biodiversity surrogates (Sarkar et al., 2006), because the availability of distributional data on natural features is generally scarce (Margules and Pressey, 2000; Crane and Bateson, 2003).

Biodiversity surrogates are hypothesised to stand for all biodiversity (Sarkar and Margules, 2002). Hence, the choice of surrogates for conservation assessments (i.e. the technical exercise to identify spatial priorities for conservation; Knight et al., 2006) is not trivial (Margules and Pressey, 2000). The number of publications reporting surrogacy performance tests certainly illustrates this concern (e.g. van Jaarsveld et al., 1998; Virolainen et al., 1999; Rodrigues et al., 2000; Williams et al., 2000; Araújo and Humphries, 2001; Garson et al., 2002; Reyers et al., 2002; Lombard et al., 2003; Sarkar et al., 2005). A wide range of features, such as species, vegetation types, species communities, environmental domains, has been used as biodiversity surrogates (Ferrier, 2002; see Grantham (2005) for a recent review).

There is, however, no consensus in the conservation community on which ones provide the best basis for achieving general biodiversity conservation (see Brooks et al. 2004c). Some authors argue that species datasets should be central to conservation assessments because species are the core component of biodiversity (Brooks et al., 2004b; Hortal and Lobo, 2006). Other authors emphasise the relevance of using surrogates such as vegetation units and environmental domains to alleviate the numerous biases of species data and/or to insure the representation and persistence of all levels of biological organisation regarded as conservation targets in their own right (Noss, 1996; Higgins et al., 2004; Molnar et al., 2004). Uncertainties on the choice of biodiversity surrogates have remained unsettled largely because they cannot be tested rigorously by empirical research (Reyers and van Jaarsveld, 2000; Sarkar et al., 2006); and many studies advocate using composite datasets in order to enhance biodiversity representation and persistence (e.g. Kiester et al., 1996; Lombard, 1997; Reyers et al., 2002; Stoms et al., 2005).

Conservation assessments are undertaken at multiple spatial scales (Driver et al., 2003;

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Sarkar et al., 2006). The notion of spatial scale encompasses the dual aspect of resolution (i.e. referring to the size of the grain) and extent (Wu and Qi, 2000). Because of its high cost per unit area, fine-scale conservation assessment (i.e. at fine resolution) is usually doable only over limited areas (i.e. in small extent) (Rouget, 2003). This shortcoming is often tackled by a strategy that consists of identifying coarse-scale priority areas over large planning domains and focusing fine-scale effort mainly within these smaller areas (Ferrier, 2002; Driver et al., 2003; Knight et al., 2006). Contrary to data on species distributions, environmental data derived from remote-sensing methodologies are easy and cheap to obtain and can be mapped consistently over extended space (Pressey and Logan, 1995; Margules and Pressey, 2000). Then again, the hierarchical nature of biodiversity (Franklin, 1993; Wu, 1999) supports the use of hierarchical protocols to the setting of priority areas, with data layers developed for all the levels of biodiversity (from species to ecosystems) (Fairbanks and Benn, 2000; Poiani et al., 2000; Rouget, 2003). Hence, the spatial scale of conservation assessments should influence the choice of biodiversity surrogates.

In addition to biodiversity surrogates, the assessment of conservation priorities must also be integrated with socio-economic considerations (Mace et al., 2000). The rationales for this is that areas with higher vulnerability (i.e. likelihood or imminence of further alteration) require more pressing conservation action (Pressey and Taffs, 2001; Wilson et al., 2005) and that the implementation success of priority areas can be improved by enhancing cost-effectiveness and by alleviating spatial conflict with human societies (Luck et al., 2003; Wessels et al., 2003). Such socio-economic criteria can take the form of spatially explicit information such as data on human population density, grazing impact and conservation costs (Balmford et al., 2000; Noss et al., 2002; Wilson et al., 2005).

The objective of this literature review is twofold. On one hand, it aims to provide a systematic and quantitative overview on the choice of biodiversity surrogates and socio-economic criteria in conservation assessments (e.g. of qualitative reviews on biodiversity surrogates: Ferrier, 2002; Gratham, 2006; Moreno and Sánchez-Rojas, 2007). On the second hand, it aims to report the effect of spatial scale on this choice. It was hypothesised that biodiversity is assessed in its hierarchical form and that coarse and/or large scale conservation assessments are generally correlated with ecosystem-type biodiversity surrogates, while fine and/or small scale ones are to species data.

2. Methods

2.1. Sampling the literature

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This review was based on search results from the ISI Web of Science database, for the period 1998 to 2005. Phrases commonly associated to conservation assessments (e.g. “area selection”, “priority areas”, Appendix 1) were entered as search topics. The search results contained 476 references. Egoh et al. (2007) used the same list of references for a review of the integration of ecosystem services in conservation planning.

A sample of 100 conservation assessments (sensu Knight et al. (2006)) was drawn for the analysis. We designed a stratified random sample of the 476 references according to five levels of geopolitical extent of planning domains:

1. global;
2. continental;
3. regional[1];
4. national;
5. subnational[2].

Conservation assessments were excluded if (i) they consisted of conceptual papers (e.g. Sarkar and Margules (2002)); (ii) the assessment was performed on purely theoretical data (e.g. Williams and ReVelle (1998)); (iii) full text references could not be obtained (five cases); and (iv) the same planning domain was assessed with the same set of data across assessments (e.g. Rodrigues and Gaston (2002)); Gaston and Rodrigues (2003)). In the last case, only one of the publications was surveyed. Where the same reference applied several approaches to the identification of priority areas in its planning domain of interest, these approaches were considered as forming part the same conservation assessment (often the case for references reporting surrogacy analyses). Two references (Hull et al., 1998; Harris et al., 2005) assessed more than one planning domain, therefore, the number of assessments outnumbers the number of references surveyed by three (i.e. 100 assessments from 97 references).

The numbers of references were too few on the global, continental and regional scales to reach the desired 20 assessments per stratum. Hence, the global and continental strata were merged into one, all references on regional scale were surveyed and the proportions for the national and sub-national scales were evenly increased. All readjustments were done provided the assessments qualified to the above rules.

[1] The regional scale is relevant to trans-border planning domains between at least two countries, but that do not fall in the continental scale.
[2] The subnational scale is relevant to planning domains strictly contained within a single country.

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Table 1 – Number of assessments per scale of each scale variable (i.e. geopolitical extent, grain and area) analysed in this literature review. A total of 100 conservation assessments (97 references) were surveyed.

Geopolitical extent		**Grain**		**Area (10^3 sq km)**	
Subnational:	30	< QDS:	37	< 100:	24
National:	29	QDS-DS:	32	100-1 000:	27
Regional:	28	≥ DS:	24	1 000-10 000:	24
Global & continental:	13	*No planning unit:*	*7*	≥ 10 000:	25

The final sample consisted of 13 assessments at global and continental scale, 28 at regional scale, 29 at national scale and 30 at sub-national scale (Table 1).

2.2. Surveying the variables

Two sets of variables were systematically surveyed.

1. Variables related to the biodiversity surrogates and socio-economic criteria considered in the conservation assessment.
2. Variables relative to two other spatial scale variables of the conservation assessment, i.e. the grain and the area.

2.2.1. Biodiversity surrogates and socio-economic criteria

There is no standardised classification of biodiversity surrogates. Three broad surrogate classes were here distinguished (Fig. 1):

1. taxonomic surrogates, based on biotic data only and defined at any level of the taxonomic hierarchy (i.e. species, genus, family);
2. environmental surrogates, based on abiotic data only;
3. assemblage surrogates consisted of community surrogates, based on biotic data only, and habitat surrogate, based on a combination of biotic and abiotic data.

The distinction between 1 and 3 above is that a taxonomic surrogate is a taxon targeted individually (e.g. three taxa A, B and C have individual representation targets; they are three distinctive taxonomic surrogates), while an community/assemblage surrogate is a group of taxa targeted collectively (e.g. A, B and C have a collective representation target; the three taxa form one community surrogate).

Surrogates for large-scale ecological and evolutionary processes are usually mapped on

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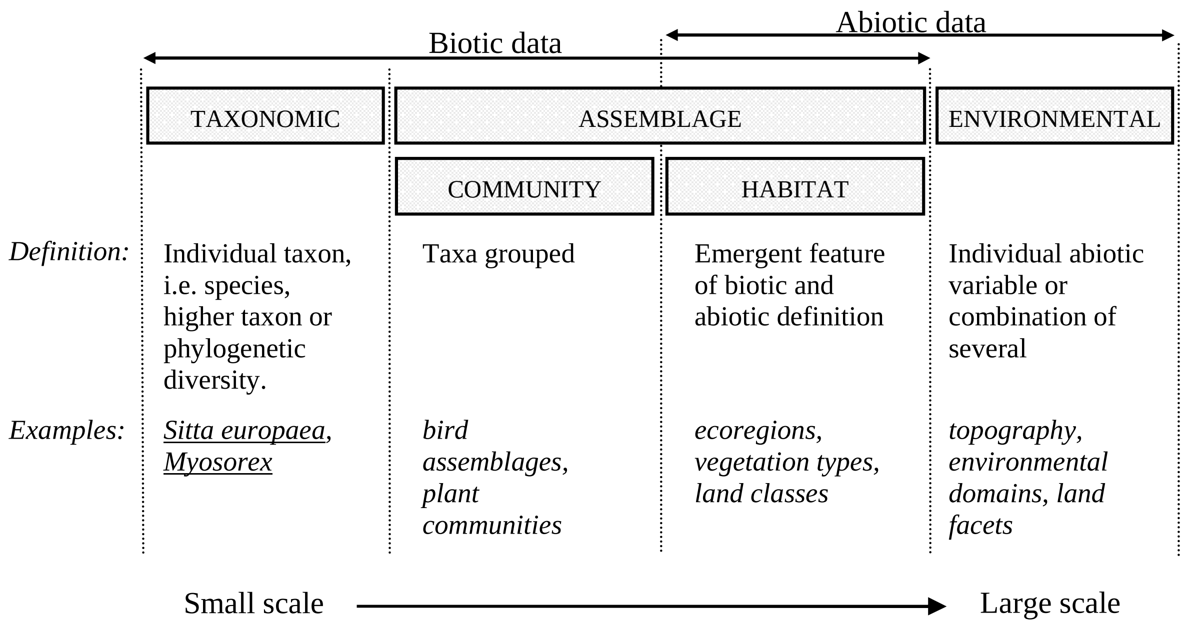

Fig. 1 – Classification scheme of biodiversity surrogates used for this literature review. Three main classes, i.e. taxonomic, assemblage and environmental surrogates, and two sub-classes for assemblage surrogates, i.e. community and habitat surrogates, were distinguished according to whether they were mapped on biotic and/or abiotic information. The hypothesis that biodiversity is assessed in its hierarchical form (see introduction for details) and that is tested here is indicated by an arrow at the bottom of the figure.

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remotely-sensed data that delineate their spatial components (Cowling et al., 1999; Rouget et al., 2003). So processes would be recorded as environmental, or possibly habitat, surrogates in this review.

Last, this classification scheme is exclusive, so one surrogate was recorded in one class only. The use of socio-economic criteria (e.g. population density, habitat transformation) was recorded.

For each assessment, the following information was recorded:

1. the taxa represented by the taxonomic and community surrogates (i.e. birds, mammals, plants, amphibians/reptiles, insects, arachnids, fungi/lichens, molluscs and fish);
2. the taxonomic level of the taxonomic surrogates;
3. the use of a measure of phylogenetic diversity for taxonomic surrogates;
4. the conservation status (i.e rare/endemic or threatened status);
5. type of indicator taxa (i.e. umbrella or flagship species);
6. type of environmental surrogate (i.e. climatic, topographic, edaphic/geologic or a combination);

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7. type of socio-economic criteria (i.e. land use/cover, human population density or other socio-economic attributes such land cost).

2.2.2. Spatial scale variables

a) Spatial extent: geopolitical extent and area

In addition to geopolitical extent the approximate area of planning domains in square kilometres was recorded. Four classes were defined based on the following categories (the breaks are arbitrary):

1. less than 100 000 sq km;
2. between 100 000 and 1 000 000 sq km;
3. between 1 000 000 and 10 000 000 sq km; and
4. equal to or more than 10 000 000 sq km.

b) Spatial grain

Most systematic conservation assessments rely on planning units. These consist of natural, administrative or arbitrary subdivisions of planning domains that differ widely in size between assessments and within regions and that are used as the building blocks of priority areas (Pressey and Logan, 1998). The average size of its planning units was used as an indication of the resolution of each conservation assessment. In the absence of planning units, the information on resolution was not recorded. Two recurrent planning unit sizes, i.e. a quarter degree square (QDS) and a degree square (DS; i.e. ~ 110 x 110 sq km at the Equator and ~ 45 x 45 sq km at polar circles), were used as breaks between three classes of grains to form roughly three levels of resolution of conservation assessments. These were:

1. less than a QDS;
2. between QDS and a DS; and
3. equal to or more than a DS.

2.3. Data analysis

A preliminary step consisted of assessing whether the stratified sample only could be used for all analyses. Thus, the effect of the stratification on the geopolitical extent was assessed by testing the statistical difference between the stratified sample and a random sample (Chi-square tests and Fisher’s exact tests, R Development Core Team, 2007). The comparison was performed on the frequency distributions of all variables, except phylogenetic diversity and indicators that had too few numbers. The random sample was a

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draw of the 100 first assessments of the search results ranked in the same random order as when drawing the stratified sample. No significant difference was found (Chi-square tests and Fisher's exact tests, $p > 5\%$), so the stratified sample was used in all analyses.

Trends per scale were explored by Correspondence Analysis (Chessel et al., 2004; R Development Core Team, 2007) and statistically assessed and compared to the overall trends by means of Fisher's exact tests (R Development Core Team, 2007).

3. Results

3.1. Overall trends

3.1.1. Biodiversity surrogates and socio-economic criteria

There were considerable differences in the use of the three classes of biodiversity surrogates. Taxonomic surrogates were by far the most commonly used surrogates (72% assessments, Fig. 2) while assemblage and environmental surrogates were respectively the second and least used surrogates. Assemblage surrogates were used in 31% of the assessments and consisted in equal proportions of community and habitat surrogates (18 assessments each). Surprisingly, environmental surrogates were used in only 11% of the assessments. In general, they were features defined on a combination of environmental factors (n=7), some climatic, topographic

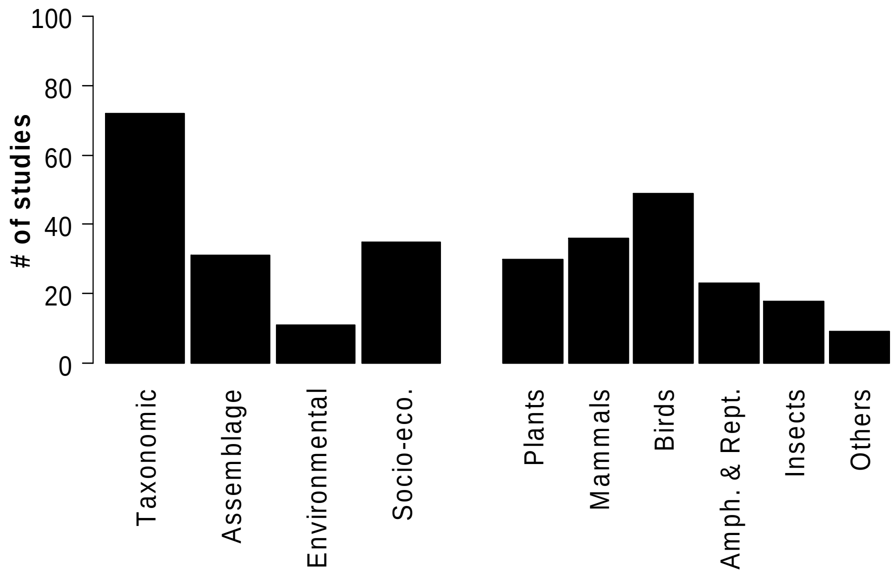

Fig. 2 – Distribution frequencies of the three biodiversity surrogate types (Taxonomic, Assemblage and Environmental) and the socio-economic criteria (Socio-eco) (left) and of the taxonomic groups represented by taxonomic and community/assemblage surrogates (right).

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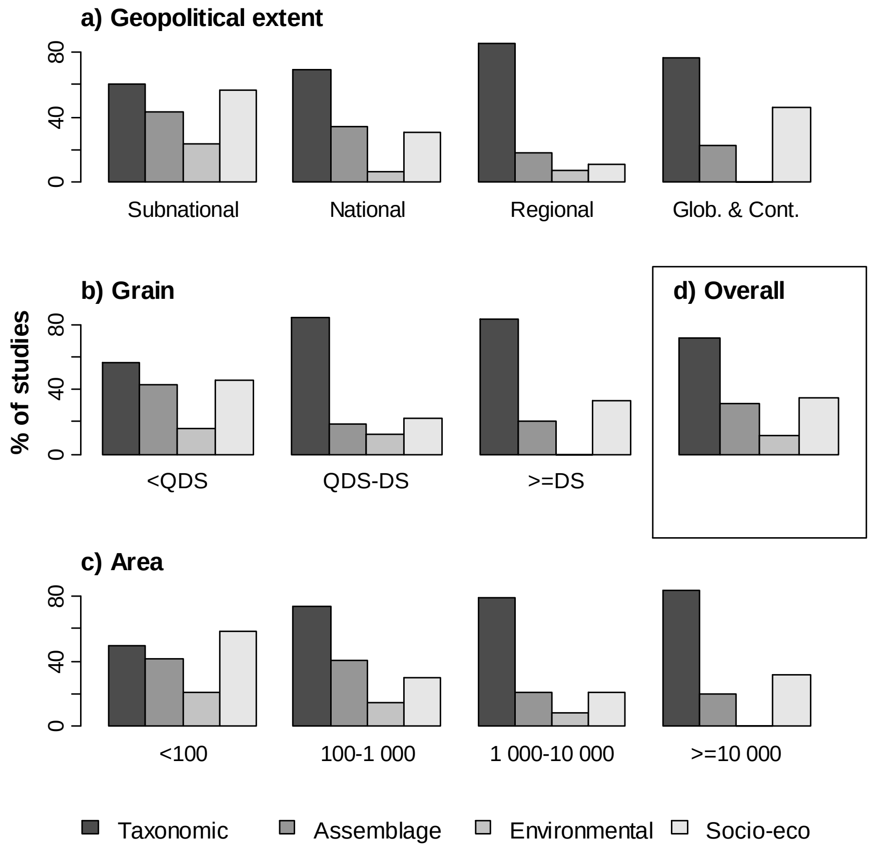

Fig. 3 – Distribution frequencies of the three biodiversity surrogate types (Taxonomic, Assemblage and Environmental) and the socio-economic criteria (Socio-eco) per scale, shown for the three scale variables, i.e. geopolitical extent (a), grain (b) and area (c). Area is indicated in thousands of sq km. Distribution frequencies on the overall sample are reported in the frame for comparison.

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and edaphic/geologic factors being used on their own in none, four and one assessments, respectively. So, in total, biodiversity surrogates defined on some abiotic variables (i.e. habitat and environmental surrogates) were found in less than 30% of the assessments.

Most assessments focused on one type of biodiversity surrogates only: 62 used taxonomic surrogates alone; 20, assemblage surrogates alone and four, environmental surrogates alone. Only 12 assessments were based on at least two biodiversity surrogate types.

The use of socio-economic criteria was reported in 35% of the assessments (Fig. 2). Socio-economic criteria consisted mostly of information on land use and cover (n=26). Measures of human population density and other socio-economic attributes were assessed in only 10 assessments each, often combined with land use and cover information.

Given the little use of assemblage and environmental surrogates, a post-result analysis consisted in tracking if trends in the use of data changed over time. All conservation

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assessments were grouped by year of publication, for the periods (i) 1998-2000, (ii) 2000-2003 and (iii) 2004-2005 to obtain three sub-samples of approximate equal size. No major differences were detected in the frequency distributions per surrogate variables. Only community surrogates were increasingly used with time, but this is for a total of just 18 assessments. Therefore, the use of assemblage and environmental surrogates remained stable over the period 1998-2005.

3.1.2. Taxonomic level and group, indicator taxa and conservation status

As a rule, taxonomic surrogates represented the species level; only seven assessments represented the genus and/or family levels. A measure of phylogenetic diversity was used in eight assessments, three being from the same reference (Hull et al., 1998).

The representation of taxonomic groups was highly uneven. Birds were surrogates in 57% (n=49) of the 86 assessments using a taxonomic or a community surrogate (Fig. 2). To a

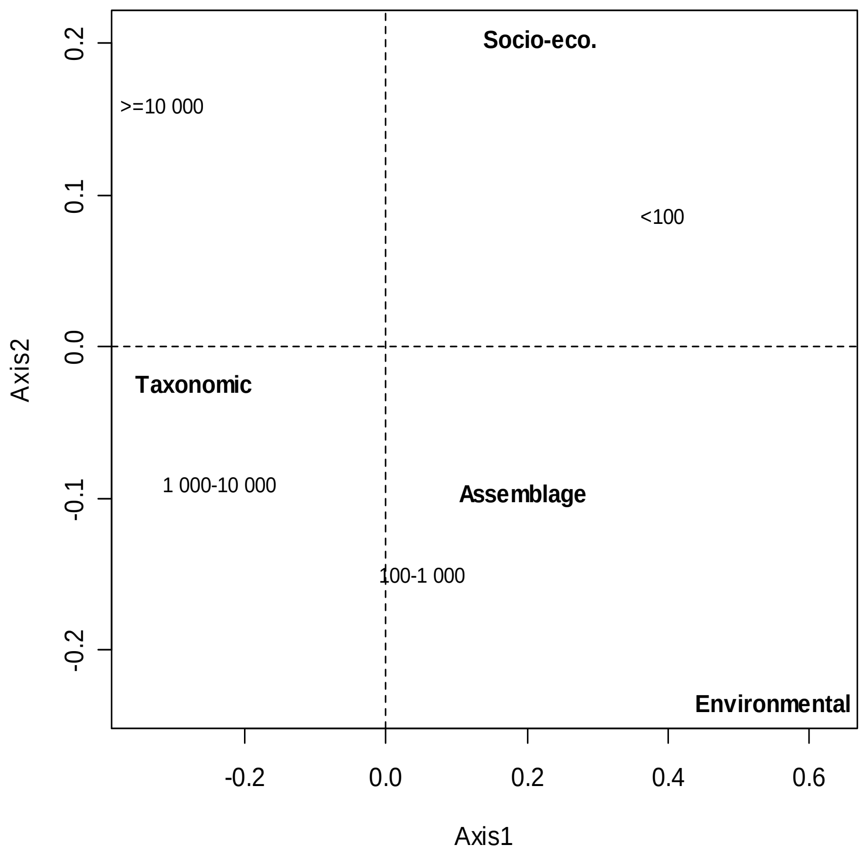

Fig. 4 – Correspondence Analysis (Chessel et al., 2004; R Development Core Team, 2007) on the three biodiversity surrogate types (Taxonomic, Assemblage and Environmental) and the socio-economic criteria (Socio-eco) shown for the four scales (< 100; 100-1 000; 1 000-10 000; ≥ 10 000 in 10^3 sq km) of the scale variable area.

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lesser extent, mammals (n=36) and plants (n=30, Fig. 2) were represented while amphibians/reptiles and insects were surrogates in about a quarter of these assessments. The other taxonomic groups were poorly represented (Fig. 2). In the majority of cases (n=51),only one taxonomic group was represented, with plants being the group most often used alone (n=15). Only, seven of these single taxon assessments were complemented by the use of environmental or habitat surrogates. Assessments relying on three or more taxa were common (n=28), but were recurrently based on the combination mammals, birds and amphibians/reptiles (n=11).

A conservation status was mentioned in more than a third of assessments using taxonomic or community surrogates (n=31), with the criteria of endemicity/rarity more often used than a threatened status (n=24 and n=15, respectively). The use of indicator taxa was much less common (only six assessments).

3.2. Trends across spatial scales

3.2.1. Biodiversity surrogates and socio-economic criteria

Surprisingly, spatial scale had a weak effect on the use of biodiversity surrogates and/or socio-economic criteria in conservation assessments. The grain was the only scale variable having an overall statistically significant effect on the use of surrogates and socio-economic criteria (Fisher's exact test, p-values < 5%); however, when compared pair-wise, frequency distributions of each grain (i.e. (i) < QDS, (ii) ≥ QDS and < DS and (iii) ≤ DS) were not statistically different one to another and neither to the frequency distributions on the overall sample (Fisher's exact test, p-values > 1%). Only the subnational and regional extents, and the areas < 100 000 sq km and ≥ 10 000 000 sq km were statistically different one to another (Fisher's exact test, p-values < 1%).

Broadly, however, taxonomic surrogates tended to be more correlated with large scale and environmental surrogates with small scales (Fig. 4), inverse to the scale hypothesis tested here. Most interestingly, no environmental surrogates were used at the three largest scales (i.e. global and continental, ≥ DS and ≥ 10 000 000 sq km). Taxonomic surrogates were assessed in ≤ 60% of the assessments at the smallest scales (i.e. subnational, < QDS and < 100 000 sq km) while they were used in ≥ 74% of the assessments at all other scales, except at national extent (69%). Inversely, assemblage and environmental surrogates were used at their highest proportions at the smallest scales (≥ 42% and ≥ 16% of assessments, respectively; Fig. 3). This was also the case for socio-economic criteria (≥ 46% of assessments; Fig. 3), in particular, with land use and cover data used in 37% of assessments with a grain < QDS and

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50% of assessments at subnational extent and of area < 100 000 sq km. Last, the use of a combination of at least two of the three types of biodiversity surrogates was largely associated to the smallest scales.

3.2.2. Taxonomic group and conservation status

None of the scale variables had an overall statistically significant effect on the choice of taxa in conservation assessments (Fisher's exact test, p-values > 5%). Distribution frequencies of taxa for any scale were not statistically different to the overall sample nor to one another (Fisher's exact test, p-values > 1%) (Fig. 5).

Birds were always the most or second most assessed taxonomic group at all scales, being used at minimum in 33% of assessments for a given scale, but tended to be more particularly assessed in planning domains with a large area (Fig. 5, area) and in conservation assessments using planning units of large average size (Fig. 5, grain). The use of data on mammals also

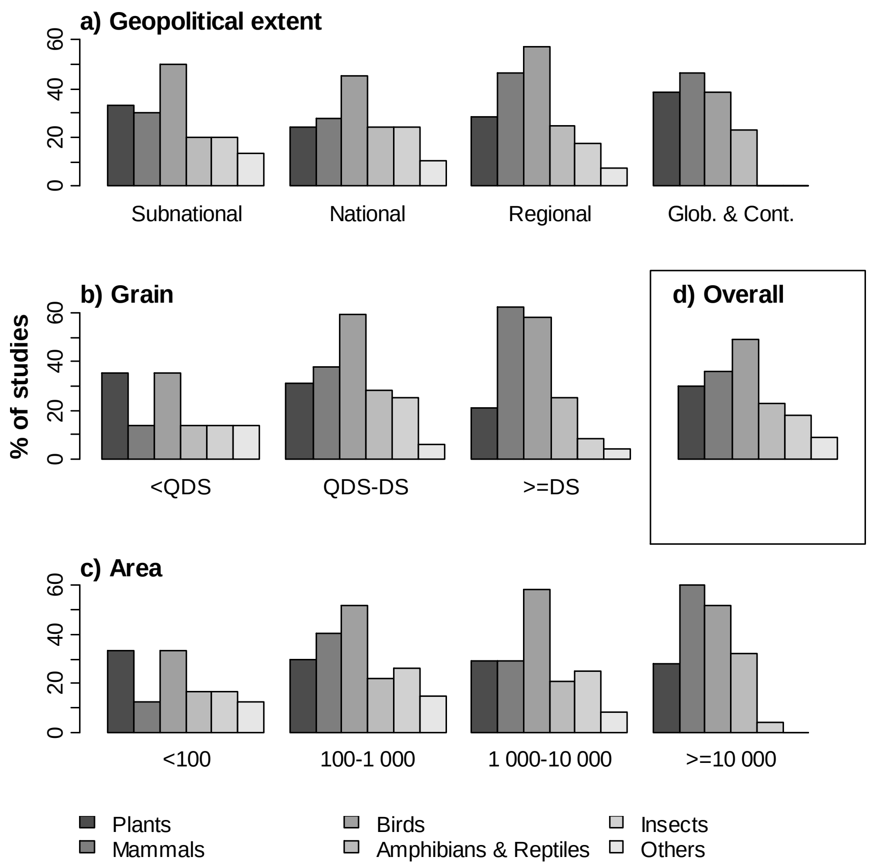

Fig. 5 – Distribution frequencies of the taxonomic groups represented by taxonomic and community surrogates per scale, shown for the three scale variables, i.e. geopolitical extent (a), grain (b) and area (c). Area is indicated in thousands of sq km. Distribution frequencies on the overall sample are reported in the frame for comparison.

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tended to be correlated with larger scales, as may be the case of amphibians and reptiles, but this is less obvious because of smaller sample size (Fig. 5). In contrast, plants were used in almost similar proportions across all scales, while insects and the very few other taxa assessed tended to be in higher proportions at smaller scales. Assessments of at least two taxa were coarsely in equal proportions at all scales.

The conservation status of taxonomic or community surrogates was mentioned more frequently at the three largest scales (in at least 52% of assessments at global and continental, ≥ DS and ≥ 10 000 000 sq km, and in less than 25% otherwise, except at regional scale (32%)).

4. Discussion

4.1. The use of biodiversity surrogates and socio-economic criteria in conservation assessments

The findings of this literature review highlighted tremendous biases in the use of biodiversity surrogates in conservation assessments. Assessments were based predominantly, and, in most cases, exclusively, on taxonomic surrogates, species being the biodiversity entity of prime interest. The bias in the use of species data towards vertebrates, mainly birds (Brooks et al., 2004a), and vascular plants (Crane and Bateson, 2003; Pressey, 2004) was also plainly apparent. The most surprising finding, however, was that biodiversity surrogates derived partly or entirely on abiotic data (i.e. habitat and environmental surrogates) were used in as few as < 30% of the assessments. These are usually remotely-sensed data that are assumed to be widely used in conservation assessments (Lombard et al., 2003; Ferrier et al., 2004). In addition, awareness in the use of surrogates other than species started in the early 1990s (Pioani et al., 2000). It is possible that the selection filter of how works and findings are published in the scientific literature has biased the survey. If such was the case, this review would highlight a communication gap.

The drawbacks of basing conservation assessments mostly on species data have been frequently discussed in the conservation literature. Not only is their value as efficient biodiversity surrogates still largely equivocal (Ferrier, 2002), but species data have well-known flaws such as geographic sampling bias, false-presence/false-absence and age of the records that impair their distribution accuracy (Maddock and Du Plessis, 1999; Pressey, 2004). The finding that half the assessments were based on single taxon approaches, when they should ideally represent several taxa (Mace et al., 2000), highlights that conservation

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priority areas may poorly represent general biodiversity. And, while composite datasets of several types of biodiversity surrogates can insure an overall better representation of biodiversity (Cowling et al., 2004; Pressey, 2004), their overall use was very low (12% of assessments and, in particular, only seven of the single-taxon assessments were complemented by the use of environmental or habitat surrogates).

One promising finding was that socio-economic criteria were used in approximately one third of the assessments. While this is a relatively low proportion, incorporation of socio-economic criteria in conservation planning has only started in recent years (Mace et al., 2000). Data consisted mainly of land use and cover data and the usefulness of other socio-economic parameters (e.g. housing density, land price) into conservation assessments needs to be more widely investigated (Naidoo et al., 2006).

4.2. The use of biodiversity surrogates and socio-economic criteria across spatial scales in conservation assessments

Spatial scale had a weak effect on the choice of biodiversity surrogates in conservation assessments. Overall, the choice of biodiversity surrogates across spatial scales followed very similar patterns, except, to some extent, at a fine scale. The hypothesis that the hierarchical organisation of biodiversity (Franklin, 1993) drives this choice was not supported by our survey of the published literature. Neither the spatial hierarchy (i.e. from populations to ecosystems) nor the taxonomic hierarchy (i.e. from species to higher taxonomic levels) of biodiversity (Sarkar, 2002) appear to influence the choice of surrogates at different scales. On the contrary, the reverse tendency was observed on the overall correlation of surrogates with scales. Assessments at intermediate and large scales were based almost exclusively on species data, most frequently of vertebrates. Fine-scale assessments relied more on the other types of biodiversity surrogates, while using less taxonomic surrogates, and also used more combinations of two or the three types of biodiversity surrogates. Half of the assessments at subnational extent and in areas < 100 000 sq km, were also based on socio-economic criteria. This is a positive finding since that fine-scale assessments are performed to inform land use decision making (Driver et al., 2003).

The coarseness of available species data implies that they are suitable for large scale assessments while assemblage and environmental surrogates are used for fine-scale planning to compensate for the lack of fine-resolution species distribution (Margules and Pressey, 2000; Ferrier, 2002; Cowling et al., 2004). Nonetheless it was striking that no or few habitat and environmental surrogates were assessed at large scales, given that remotely-sensed data

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can be mapped with consistency over large areas and are relatively rapid and cheap to obtain (Margules and Pressey, 2000; Ferrier, 2002; Sarkar et al., 2005) and their use appears particularly suitable for the assessment of large planning domains.

The assumption that species data provide good quality approximations for the representation and persistence of other levels of biodiversity may be false (Conroy and Noon, 1996; and see Chapter 3). The identification of large-scale priority areas based on species data can have two undesirable outcomes. First, the focus on species means that the representation and persistence of higher levels of the biodiversity hierarchy is not explicitly addressed. These levels may therefore lack adequate conservation. In addition, given the complex relationships that link biological levels one to another, the disruption of function at one level, may have unexpected consequences on persistence at other levels (Wu, 1999). Second, given that priority-area setting may be addressed by hierarchical top-down protocols (Ferrier, 2002; Driver et al., 2003; Cowling et al., 2004; Knight et al., 2006), where large-scale priority areas are missed, all the potential fine-scale priority areas falling within their boundaries may also be missed. This is very likely to happen where large-scale priority areas are identified for the representation of a few species taxa. In both cases, the goal of representation and persistence of as many natural features as possible is not attainable (Pressey et al., 1993; Cowling et al., 1999; Margules and Pressey, 2000).

5. Conclusion

This review highlights the opportunistic use of data in large-scale conservation assessments. It also demonstrates that, even given the expediency of their mapping, remotely-sensed data are not as widely used as were first expected. It is, however, crucial that research and technical applications on the use of assemblage and environmental surrogates be more widely shared, because conservation assessments at intermediate and large scales need to be more robust and based widely on these types of data.

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Appendix A. Search terms used for the compilation of the ISI journal references in Web of Science database (Egoh et al., 2007)

"area selection", "area identification", "bioregional conservation", "bioregional conservation planning", "bioregional planning", "conservation assessment", "conservation evaluation", "conservation planning", "conservation value", "ecoregional assessment", "ecoregional conservation", "ecoregional conservation planning", "integrated conservation planning", "natural areas identification", "natural areas planning", "natural areas selection", "natural areas selection", "reserve selection", "priority areas", "priority conservation areas", "priority sites", "reserve planning", "reserve selection", "reserve site selection", "site identification", "site selection" and "spatial conservation".

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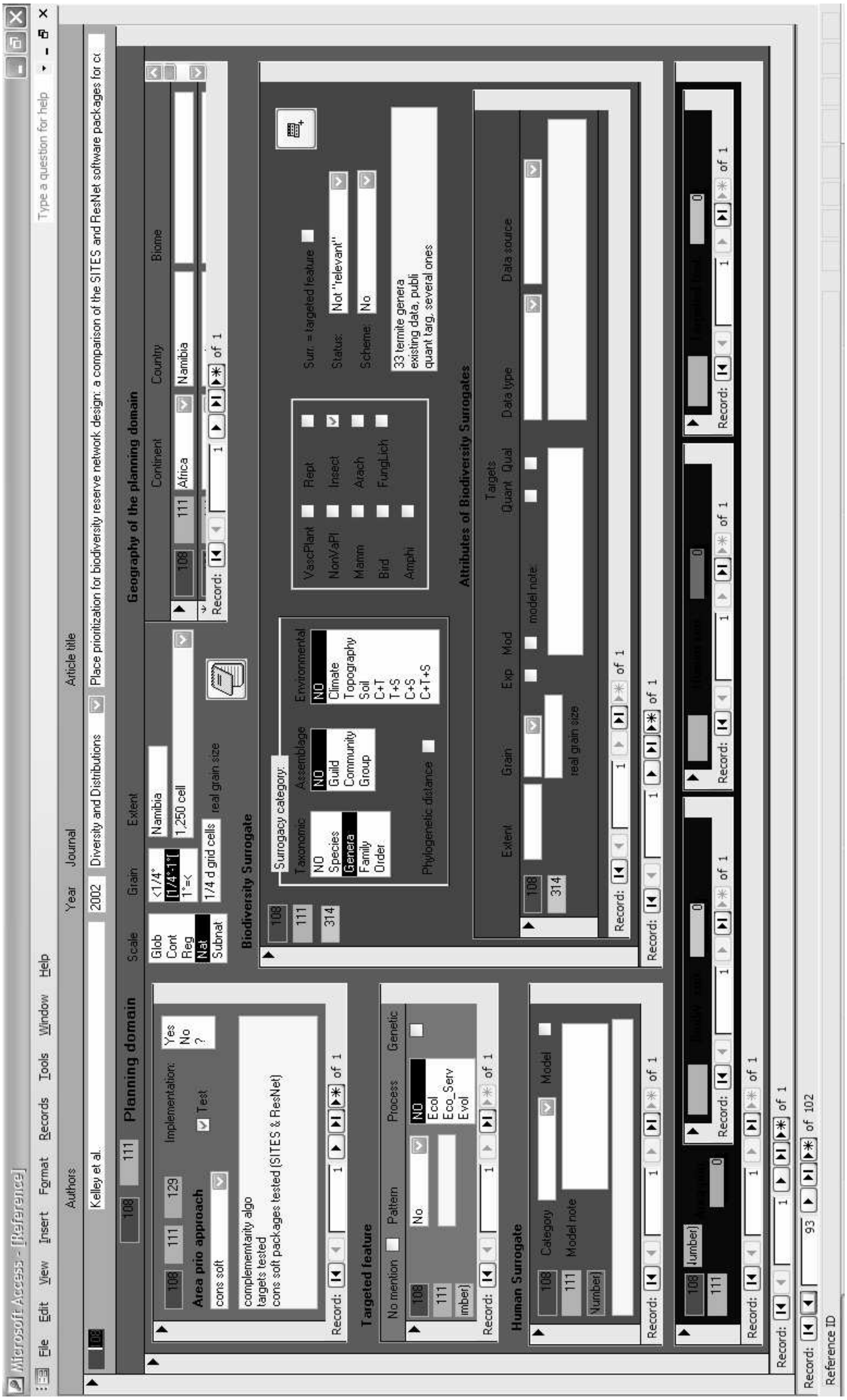
Appendix C. Microsoft Access database used for the surveying.



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Chapter 3

The effectiveness of regional-scale biodiversity surrogates at representing local-scale biodiversity surrogates in Réunion Island

A B S T R A C T

The allocation of conservation efforts needs to be optimised because resources are scarce. This study assessed the complementarity between regional- and local-scale assessments and the implications on the choice of biodiversity features at both scales on Réunion Island. A biodiversity assessment was performed at a regional scale and measured against a finer-scale assessment performed over a smaller planning domain. Two datasets composed of species distributions, habitat patterns and spatial components of ecological and evolutionary processes were compiled as biodiversity surrogates at each scale. The regional-scale surrogates were tested in terms of incidental representation and correlation of irreplaceability patterns of complementarity-based area selections, for the local-scale biodiversity surrogates. Targets for processes were never met incidentally, while threatened species and fragmented habitats were also usually missed. Requiring only 12% of the local planning domain, the regional assessment targeting species was the least effective at representing local-scale features at target level. On the contrary, the one targeting habitats achieved a significant proportion of targets (67%) and was well correlated with local-scale assessments. The results highlighted that all three types of surrogates are necessary for maximum biodiversity conservation. They further suggested that a spatial strategy based on coarse filters for regional-scale assessments, and fine filters for local-scale ones, can be an effective approach to systematic conservation assessments. The fine-scale mapping of fine filters can be principally focused on transformed habitats. Regional- and local-scale assessments if undertaken independently should not affect the efficiency of the overall conservation network significantly.

Keywords: Coarse filters; Conservation assessments; Efficiency; Fine filters; Habitat transformation; Spatial protocol

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1. Introduction

Conservation ambitiously aims to preserve all biodiversity (Noss, 1990; Sarkar, 2002). Systematic approaches have been developed to identify and implement networks of priority areas that, together, preserve a sample of all the biodiversity (Margules and Pressey, 2000). Here, I focus on the technical phase of systematic conservation planning that deals with the identification of priority areas, viz. the conservation assessment (Knight et al., 2006). Several explicit characteristics define a systematic conservation assessment (Margules and Pressey, 2000; Knight et al., 2006). Amongst others, there is the use of measurable variables that serve as surrogates for all biodiversity and help in the process of comparing areas and prioritising them based on their biological value (Margules and Pressey, 2000; Sarkar et al., 2002; Chapter 2).

The biological value of areas is derived from georeferenced and quantifiable data that are available only for a few well-documented biodiversity features (Brooks et al., 2004a; Sarkar et al., 2005). These features are assumed to provide a representative measurement of the overall biological value of the areas being assessed, and therefore act as biodiversity surrogates (Sarkar and Margules, 2002; Williams et al., 2006). Data used as biodiversity surrogates usually consist of maps of species distributions, species communities, land types, or environmental domains, (see Grantham (2005) for a review). Biodiversity surrogates can be classified in two broad categories according to the type of data they are defined upon: (i) taxonomic surrogates typically are defined on biological data and consist of species distribution or species assemblage distribution; and (ii) environmental surrogates are abiotic variables (e.g. topographic heterogeneity) or a combination of biotic and abiotic information, such as community distribution and vegetation types (Grantham, 2005; Chapter 2).

The effectiveness of a biodiversity surrogate refers to its ability to insure an adequate representation of other biodiversity features (Sarkar et al., 2005). There is, however, no consensus amongst conservationists on which best surrogates to use (see Conservation Biology, Volume 18: Brooks et al., 2004b, c; Cowling et al., 2004; Higgins et al., 2004; Molnar et al., 2004; Pressey, 2004). Some authors emphasise the relevance of using environmental surrogates to fulfil the goals of representation and persistence of biodiversity patterns and processes at all levels of biological organisation (e.g. Higgins et al., 2004; Pressey, 2004; Sarkar et al., 2006), while other authors debate that species datasets are first-order information to priority-setting because species are regarded as the core component of biodiversity (e.g. Brooks et al., 2004c; Hortal and Lobo, 2006). This issue has remained unsettled largely because it cannot be assessed rigorously by empirical research (Sarkar et al.,

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2006). There is indeed no measure of global biodiversity against which the effectiveness of biodiversity surrogates can be assessed comprehensively (Sarkar, 2002), and at best, what can be tested, is how well a set of known biodiversity features can represent another set of known biodiversity features (Sarkar and Margules, 2002). Furthermore, shortcomings of data quality and availability (Ferrier, 2002; Crane and Bateson, 2003) often make findings subject to caution (e.g. Larsen and Rahbek, 2005), while the comparison of results across regions is impossible due to the lack of site duplication and the variety of methods employed (Reyers and van Jaarsveld, 2000). Surrogacy performance tests can only provide a partial and equivocal empirical assessment of the problem of global biodiversity representation.

Nonetheless, this field of research should ultimately formulate useful recommendations to conservation planners on which data to compile depending on the time and resources they have available (Stoms et al., 2005). An aspect generally not mentioned in the literature is that the process of compiling and editing datasets on biodiversity surrogates can be a cumbersome and time-consuming enterprise. Fine-scale conservation assessment is also costly and often affordable over limited areas (Rouget, 2003), so large areas are normally assessed through coarse-scale assessments (Mittermeier et al., 1998; Ferrier, 2002; Driver et al., 2003). However, priority areas identified at a broad scale are often too coarse for implementation of conservation actions on the ground. A fine-scale assessment is usually required to refine areas of intervention. This paper focuses on the type of data to be compiled for a fine-scale assessment, so that information is not redundant from, but rather complementary to, the broad-scale assessment.

An initiative to introduce systematic conservation planning on Réunion Island (Lagabrielle, 2007) presented an opportunity to investigate this question. Two nested regions (i.e. at regional and local scale) were assessed with different combinations of datasets on terrestrial biodiversity and the extent to which area selections at regional-scale represented the local-scale biodiversity surrogates was analysed. The biodiversity surrogates consisted of an array of commonly used ones, which are species distributions, community and habitat distributions (Sarkar and Margules, 2002; Pressey, 2004), as well as spatial components of ecological and evolutionary processes (Cowling and Pressey, 2001; Rouget et al., 2003). This study attempted to draw conclusions on the complementarity of regional- and local-scale assessments and was based on the assumption that when a regional network represented local-scale biodiversity features well, the resources spent on the acquisition of the fine-scale dataset could have been spent on another element of the conservation assessment. This rule supports the increase in the optimisation of conservation effort allocation (Wilson et al., 2006).

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2. Material and methods

2.1. Study area

Réunion Island (21°S, 55°E) is a volcanic oceanic island of the Mascarene Archipelago, in the Indian Ocean (see box in Fig. 1). It is 2 512 sq km in extent and characterised by substantial environmental heterogeneity. The climate is humid tropical. The highest peak of the island culminates at 3 070 m (Piton des Neiges) in the centre and the currently active volcano culminates at 2 631 m (Piton de la Fournaise) in the South-east. The sharp relief is furrowed by deep ravines and supports a thermal altitudinal gradient (mean temperatures vary between 21°C in winter and 26°C in summer on the coast, and respectively, between 12°C and 17°C at about 1 500 m), a highly contrasting east-west rainfall gradient (mean annual rainfall is > 6 m in the East to < 1.5 m in the West; Météo France www.meteo.fr) and zones of diversified

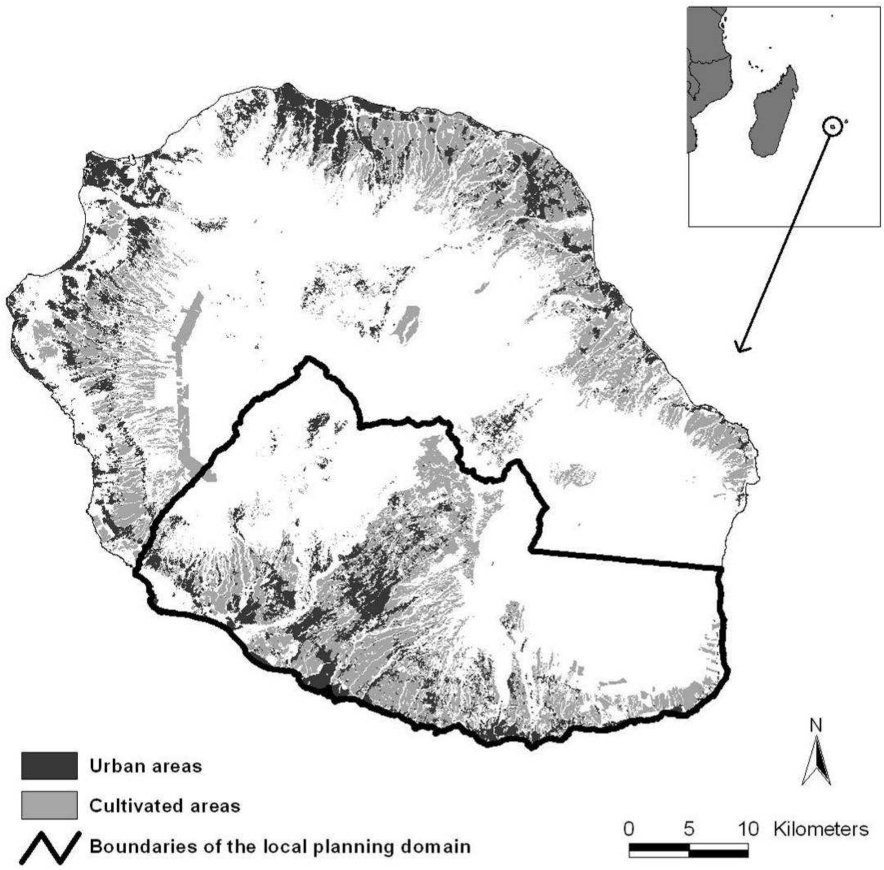

Fig. 1 – Patterns of habitat transformation in Reunion Island. Both planning domains analysed in this study are the whole island (i.e. the regional planning domain) and the area delimited by the thick black line (i.e. the local planning domain).

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microclimates.

Habitat types are structured along the altitudinal and rainfall gradient, from littoral grass prairie to alpine shrublands through evergreen forests or savannas (Cadet, 1980). Plant species richness is relatively low but the level of endemism is high (Le Corre and Safford, 2001; http://flore.cbnm.org). The flora of the island counts a high proportion of orchids whose populations are also largely affected by the environmental conditions associated wit altitude (Jacquemyn et al., 2005). Two endemic seabirds are known to breed only on the island (Le Corre et al., 2002). Réunion Island has been identified as part of the Madagascar and islands biodiversity hotspot for its high endemism and habitat loss (Myers et al., 2000).

Habitat transformation (including alien plant invasion) on Réunion Island is estimated at 73% and is concentrated in the more accessible lowlands (Fig. 1). Invasion by alien species and land transformation by clearing and human population growth are currently very high and still increasing (Strasberg et al., 2005). The two other Mascarene islands (Mauritius and Rodrigues), with lower landscape diversity, have undergone more extensive habitat loss (Baret et al., 2006). Réunion island has retained higher populations of several endemic birds than in the other small Indian Ocean islands (Thiollay and Probst, 1999).The challenge for conservation on Réunion Island is pressing.

2.2. Study design

The estimation of the representation of the local-scale biodiversity surrogates by assessments for the regional-scale surrogates was tested in two ways (e.g. Lombard et al., 2003; Warman et al., 2004b):

1. by measuring the proportion of fine-scale biodiversity features (set B) that incidentally reached their conservation target in a near-minimum set selected for regional-scale biodiversity surrogates (set A) (Fig. 2a);
2. by analysing how the irreplaceability patterns for A were spatially correlated to those for B (Fig. 2b).

The near-minimum set method assesses the level of target achievement in one good configuration of priority areas identified amongst others; the irreplaceability method, on the other hand, assesses the probability of target achievement across the whole planning domain (Pressey et al., 1994; Lombard et al., 2003; Warman et al., 2004b).

The conservation assessments were performed according to principles of systematic conservation planning (Margules & Pressey, 2000; Knight et al., 2006). The subsequent three steps of the procedure were followed:

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1. two planning domains (local and regional) were mapped and divided up into planning units,
2. geographic information system (GIS) maps on the distribution of biodiversity surrogates were compiled and surrogates were individually assigned a quantitative conservation target, and
3. the selection of priority areas was based on the complementarity principle and performed in the conservation planning software Marxan (Ball and Possingham, 2000) and CLUZ (Smith, 2004) to represent surrogates at target level.

A robust systematic conservation planning exercise requires that existing reserves and contextual parameters, such as socio-economic variables, are taken into consideration (Margules and Pressey, 2000). Such parameters influence the selection process and would have made it difficult to distinguish between their effect and the effect of the biological data on the selection outcomes. They were therefore not included in this surrogacy analysis.

2.3. Planning domains and planning units

The small planning domain (i.e. the local scale) was delineated by municipal boundaries and consisted of approximately one third (943 sq km) of the extent of the whole island, itself being the large planning domain (i.e. the regional scale; Fig. 1).

The comparison of the biodiversity value of sites one to another is usually done by dividing planning domains into elementary building blocks called planning (or selection)

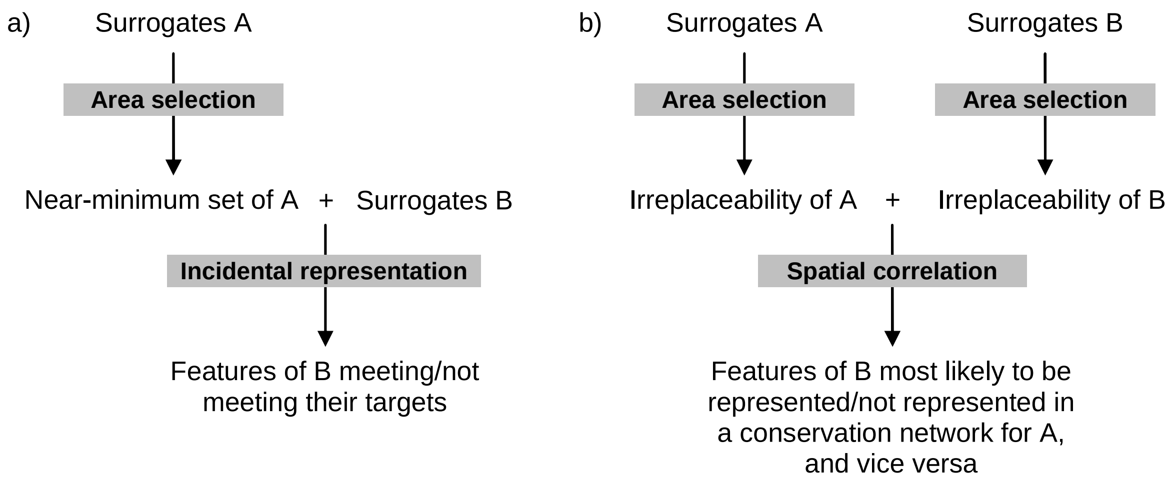

Fig. 2 – Near-minimum set (a) and irreplaceability (b) approaches used for the surrogacy analysis.

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units (Pressey and Logan, 1998). We divided each planning domain in hexagonal planning units (10.39 ha each). The hexagons were identical in size, orientation and position for both planning domains. Hence, all 9 378 units of the local scale were nested within the 24 630 units of the regional scale. Choosing different planning units would have influenced the selection process (Warman et al., 2004b; Rouget, 2003; Rodrigues and Gaston, 2001).

2.4. Biodiversity surrogates

The dataset of biodiversity surrogates for biological patterns and ecological and evolutionary processes of each scale comprised of: (i) the distribution range of natural habitat types (hereafter, habitats), (ii) the distribution of individual species or assemblages (hereafter, species; Appendix 1) and (iii) the geographic patterns of ecological and evolutionary processes (hereafter, processes; Table 1). Existing GIS maps were obtained from biodiversity organisations or new maps were generated for the project using expert knowledge (see Table 1 and Acknowledgments for the sources). The scales were approximately 1:100 000 and 1:50 000 for the regional- and local-scale datasets, respectively.

Each biodiversity surrogate was assigned a quantitative conservation target that is the minimum amount (in terms of number of occurrences or areas of land) to be included in the priority areas (Pressey et al., 2003). There is no universal and perfect recipe to the setting of conservation targets (Sarkar et al., 2006). Here, some premises of the approach of Pressey et al. (2003) were followed to define percentage targets scaled to reflect differences in apparent requirements for protection of each biodiversity feature. The targets were inferred, where possible, on considering environmental and biological heterogeneity, natural rarity and vulnerability. Higher heterogeneity, rarity and vulnerability reflected higher requirements for protection, and therefore higher conservation targets.

2.4.1. Habitats

Strasberg et al (2005) derived a map of habitats for Réunion in a two-tier classification. Habitat types were derived mainly on climate, topography and geology, using information from the literature, expert knowledge and remote-sensed data. The regional- and local-scale habitats comprised of the extant area of six and 18 habitats, respectively (Table 1).

Targets ranged from 20 to > 100% at the regional scale and 8 to > 100% of the habitats' current extent at the local scale. Regional-scale habitats were classified in two groups according to their environmental heterogeneity (i.e. factors on altitude, slope, local relief, soil and precipitation) and a percentage of 20 and 30% of estimated original extent was attributed

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Table 1 – Biodiversity surrogates used at regional and local scales

Regional-scale surrogates		**Targets in percent of current extent**
Habitats	6 land types (Strasberg et al., 2005)	20 to >100%
Species	8 threatened vascular plant species (CBNM[a])	100%
	16 vertebrate species or species assemblages (SEOR[b], Nature et Patrimoine[c])	10, 20 or 60%
Processes	5 processes (Lagabrielle, 2007)	100%
Local-scale surrogates		
Habitats	18 land types (Strasberg et al., 2005)	8 to >100%
Species	19 vascular plant species or species assemblages (E. Rivière[d])	20, 30, 40 or 100%
Processes	6 processes (Lagabrielle, 2007)	100%

[a] Conservatoire Botanique National des Mascarins
[b] Société d'Etudes Ornithologiques de la Réunion
[c] Nature et Patrimoine
[d] CIRAD, chemin de l'Irat, 97410 Saint-Pierre, Réunion, riviere@cirad.fr

respectively to the least (n=2) and the most (n=3) heterogeneous habitats (Lagabrielle, 2007). Local-scale habitats were classified in quartiles according to their environmental heterogeneity (same as above), level of plant species endemism and level of plant species richness (Lagabrielle, 2007). A percentage of 10, 20 and 30% of estimated original extent was assigned respectively to the habitats falling in the lowest (n=4), both intermediate (n=8) and the highest (n=5) scoring quartiles of the three characters. An exception was made for regional- and local-scale wetlands that were assigned a target of 100%, based on the assumption that their sustainability relied on their maximum integrity. Percentages were converted into areas calculated from the original extents of the habitats.

2.4.2. Species

All species were indigenous and some strictly endemic to the island. The regional-scale dataset consisted of both plant and vertebrate species (Table 1). Data on plants were locality records of eight threatened species, whose threat status had been determined by the Conservatoire Botanique National de Mascarin (CBNM, http://flore.cbnm.org) using the same

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categories and criteria as the IUCN Red List Categories and Criteria version 3.1 (2001). Data on vertebrates comprised locality records or distribution ranges for one bat species, two reptiles, eight forest bird species, two forest bird species assemblages and nesting sites of two oceanic bird species and one oceanic bird species assemblage. The bat species, both oceanic birds and two forest bird species were IUCN Red List threatened species (IUCN Red List version 3.1, 2001).

The local-scale dataset consisted of the distribution ranges of 19 plant species or plant species assemblages or communities (Table 1) mapped on local expert knowledge. Two of these species was listed threatened by the IUCN (Red List version 2.3, 1994) and five other species were assessed threatened by the CBNM (http://flore.cbnm.org). Three CBNM threatened species (*Delosperma napiforme*, *Chamaesyce viridula* and *Pemphis acidula*) were also included in the regional-scale dataset.

Species targets were calculated as a proportion of their known current distribution. Targets for plants took the values 100% at regional scale and 20% (n=9), 30% (n=2), 40% (n=1) and 100% (n=7) at local scale (Table 1). The percentages were attributed on criteria of rarity and vulnerability of the CBNM and the official national protection status that was also used as an indicator of protection requirement. Marginally the targets were fixed on expert opinion or on readjustment with targets from overlapping habitats. Targets for vertebrates were 20 (n=4), 40 (n=7) and 100% (n=5) of current distributions. Species with higher conservation priority defined by an association of local ornithologists (Société d'Etudes Ornithologiques de la Réunion, www.seor.fr) and/or higher IUCN threatened status (IUCN Red List Categories and Criteria version 3.1 (2001)) and/or endemic to the island (i.e. used as a surrogate for overall natural rarity) were assigned a higher conservation target.

2.4.3. Processes

Protocols have been proposed in South Africa on how large-scale processes can be integrated into conservation assessments (e.g. Cowling et al., 1999; Cowling and Pressey, 2001; Desmet et al., 2002; Cowling et al., 2003; Rouget et al., 2003). They were followed here to identify, map and select areas for the representation of their spatial components in priority areas (see Lagabrielle, 2007 for more details).

Five major spatial components of that sustain the ecological and evolutionary persistence of the insular biodiversity of Réunion were mapped at regional scale. These were lowland-upland gradients that sustain plant diversification (Warren et al., 2006) and seasonal feeding migrations of birds and insects (in Lagabrielle, 2007), oceanic-terrestrial interface that supports the feeding of marine birds nesting inland (Le Corre and Safford, 2001) and species

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colonisation (Cadet, 1980), isolated topographic boundaries for the allopatric diversification of taxa (in Lagabrielle, 2007), riverine corridors of perennial rivers that support top-down nutrient flows and bird movements (Le Corre and Safford, 2001) and regional-scale habitat interfaces for plant and animal lineages diversification (Cadet, 1980). For the local-scale assessment, the riverine corridors of the smaller non-perennial rivers were added and the regional-scale habitat interfaces were replaced by the local-scale habitat interfaces. Hence, there were five "processes" at regional scale and six at local scale, four being common to both scales (Table 1). Their extant areas were targeted at 100% of their range, based on the assumption that their functionality required their full integrity (Cowling et al., 2003).

2.5. The selection of priority areas

Networks of areas were identified in order to maximise efficiency (Pressey & Nicholls, 1989), which consisted of representing biodiversity surrogates at their target level within the smallest possible area (Pressey et al., 1993; Sarkar and Margules, 2002). This is best performed by selecting areas that are the most complementary in the features they contain (Pressey et al., 1993). Complementarity-based area selections identify solutions called minimum or near-minimum sets (Pressey et al., 1997; Pressey and Taffs, 2001). They have widely been used in real-world conservation planning (Justus and Sarkar, 2002).

We used the simulated annealing algorithm of the Marxan conservation software (Ball and Possingham, 2000), with the CLUZ interface (Smith, 2004), to identify near-minimum sets. The objective assigned to Marxan for each near-minimum set was simply to meet the targets of the chosen surrogates, since other objective rules that can be fixed in the software (e.g. parameters for cost and design criteria, Ball and Possingham, 2000) were not used here. The near-minimum sets were produced by targeting habitats, species and processes separately and collectively. The seven possible combinations at each scale were the selections targeting habitats (H), species (S), processes (P), habitats and species (HS), habitats and processes (HP), species and processes (SP), and habitats, species and processes (HSP). For each combination, a suite of 200 and 150 near-minimum sets was obtained at regional and local scales, respectively.

For the surrogacy analysis, each regional best (i.e. the most efficient) near-minimum set was used for approach A in Fig. 2. For approach B in Fig. 2, the Marxan output called the "summed solution" was used. This is the summary of the total number of times each planning unit is selected among all near-minimum sets. Hence, the summed solution tells how necessary each planning unit is to achieve conservation targets which corresponds to a

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measure of its irreplaceability (Ball and Possingham, 2000; Ferrier et al. 2000). The seven summed solutions produced at each scale were analysed.

2.6. The surrogacy analysis

The regional (best) near-minimum sets and summed solutions were clipped to the local planning domain boundaries in ArcView 3.2 (ESRI, Redlands, CA, USA). All further comments and discussion are therefore relevant to the local planning domain extent only, except if specified otherwise.

2.6.1. The near-minimum set approach

The seven regional near-minimum sets were compared one to another in terms of (i) the local-scale biodiversity surrogates incidentally represented at target level and (ii) the total area selected (Reyers and van Jaarsveld, 2000; Lombard et al., 2003). There is incidental representation when the selection based on a given biodiversity surrogate also insures the representation of non-targeted features (Warman et al., 2004a). In the case of five local-scale habitats that had targets larger than their remaining extent, targets were considered achieved when the total remaining area was selected. A good regional near-minimum set was one that achieved targets for a larger number of local-scale surrogates in a smaller total area selected,

Table 2 – Percentage of area selected per regional- and local-scale near-minimum sets relative to the local-scale planning domain

	% of area selected	
Near-minimum sets	**local scale**	**regional scale**
Habitats	22	22
Species	17	12
Processes	43	40
Habitats + species	23	25
Habitats + processes	47	43
Species + processes	44	40
Habitats + species + processes	47	43

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following the efficiency principle (Pressey & Nicholls, 1989). Local-scale features that met and did not meet their targets were also identified.

The regional near-minimum sets were also compared to random selections (e.g. Rodrigues et al., 2000) that were made for the same total area selected as obtained for the near-minimum sets. Twenty random selections were performed for each area and the one achieving the highest number of targets was used for the comparison. These comparisons indicate whether the regional conservation assessments perform better than randomly selecting units in achieving targets for local-scale biodiversity features.

2.6.2. The irreplaceability approach

The level of spatial correlation between the regional- and the local-scale irreplaceability values was assessed using a Spearman's rank correlation coefficients (R Development Core Team, 2007). The implementation of conservation networks often happens over prolonged periods and requires spatial flexibility (Pressey and Taffs, 2001; Pence et al., 2003). The

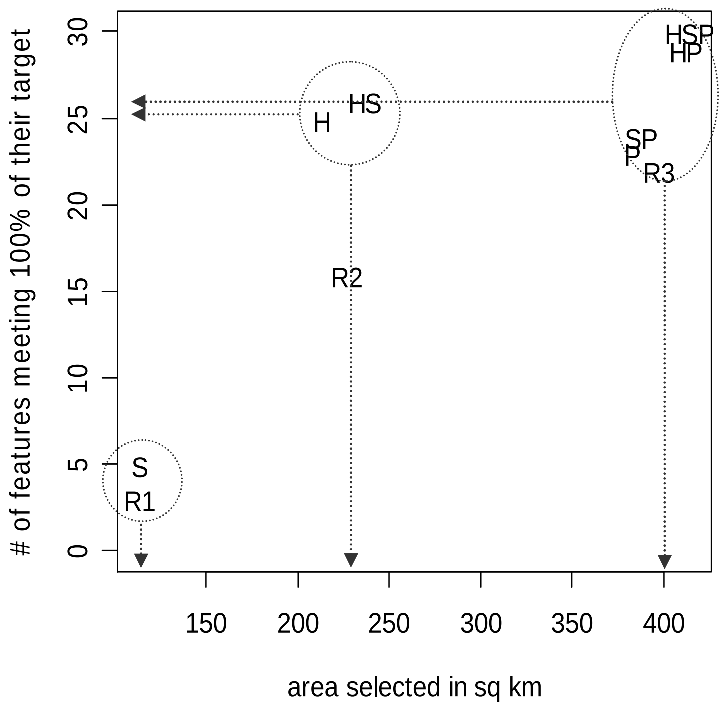

Fig. 3 – Number of local-scale species and habitats meeting their targets by incidental representation in the regional near-minimum sets (each combination is indicated by (a) letter(s) with H for habitats, S for species and P for processes). R1, R2 and R3 are the random selections (see text for explanation). Three groups are distinguishable in terms of total area selected (i.e. S, H + HS and P + SP + HP + HSP) and two in terms incidental representation at target level (i.e. S and H + HS + P + SP + HP + HSP).

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assumption here is that where patterns of irreplaceability values for one surrogate are spatially similar to patterns of irreplaceability values for another surrogate, a conservation assessment for either surrogate is likely to incidentally represent the other surrogate in a final conservation network (Warman et al., 2004b). To the contrary, if there is low or no similarity, it is likely that priority areas identified for both surrogates would complement each other (Margules et al., 2002) and both surrogates should be included to ensure their adequate representation in the final network.

3. Results

3.1. The near-minimum set approach

3.1.1. Incidental representation and area selected

None of the regional near-minimum sets incidentally met the targets for all local-scale biodiversity surrogates. Target achievement ranged from five (13%) to 30 (77%) features. None of the targets of processes were ever incidentally met in regional near-minimum sets[1]. The total area selected ranged from 12% to 43% of the planning domain and was comparative to the total area selected by the near-minimum sets performed on the local-scale biodiversity surrogates (Table 2).

Three groups of near-minimum sets (i.e. excluding the random selections R1, R2 and R3) were apparent on the selected area axis (x axis) and two were observed on the target achievement axis (y axis) of Fig. 3. The group that comprised only the regional near-minimum set targeting species (S) achieved both minimal incidental target achievement and smallest area selected (Fig. 3, see also Fig. 4 for a representation). Targeting regional-scale species in any of the near-minimum set combinations also only marginally improved the incidental representation of local-scale surrogates. Hence, regional-scale species were relatively inefficient surrogates for local-scale biodiversity features.

The near-minimum set combinations targeting regional-scale processes (P, HP, SP and HSP) achieved relatively high incidental target achievement for both local-scale species (58 to 79% of them reaching their targets) and habitats (67 and 83%) but were also the most land-hungry selections (Fig. 3, see also Fig. 4), and required between 40 and 43% of the planning domain (> 400 sq km, Fig. 3). The regional near-minimum sets targeting habitats (H)

[1] Only two of the local-scale processes were different from the regional-scale ones. Therefore, the actual incidental representation was for these two processes only when regional processes were targeted in regional-scale assessments.

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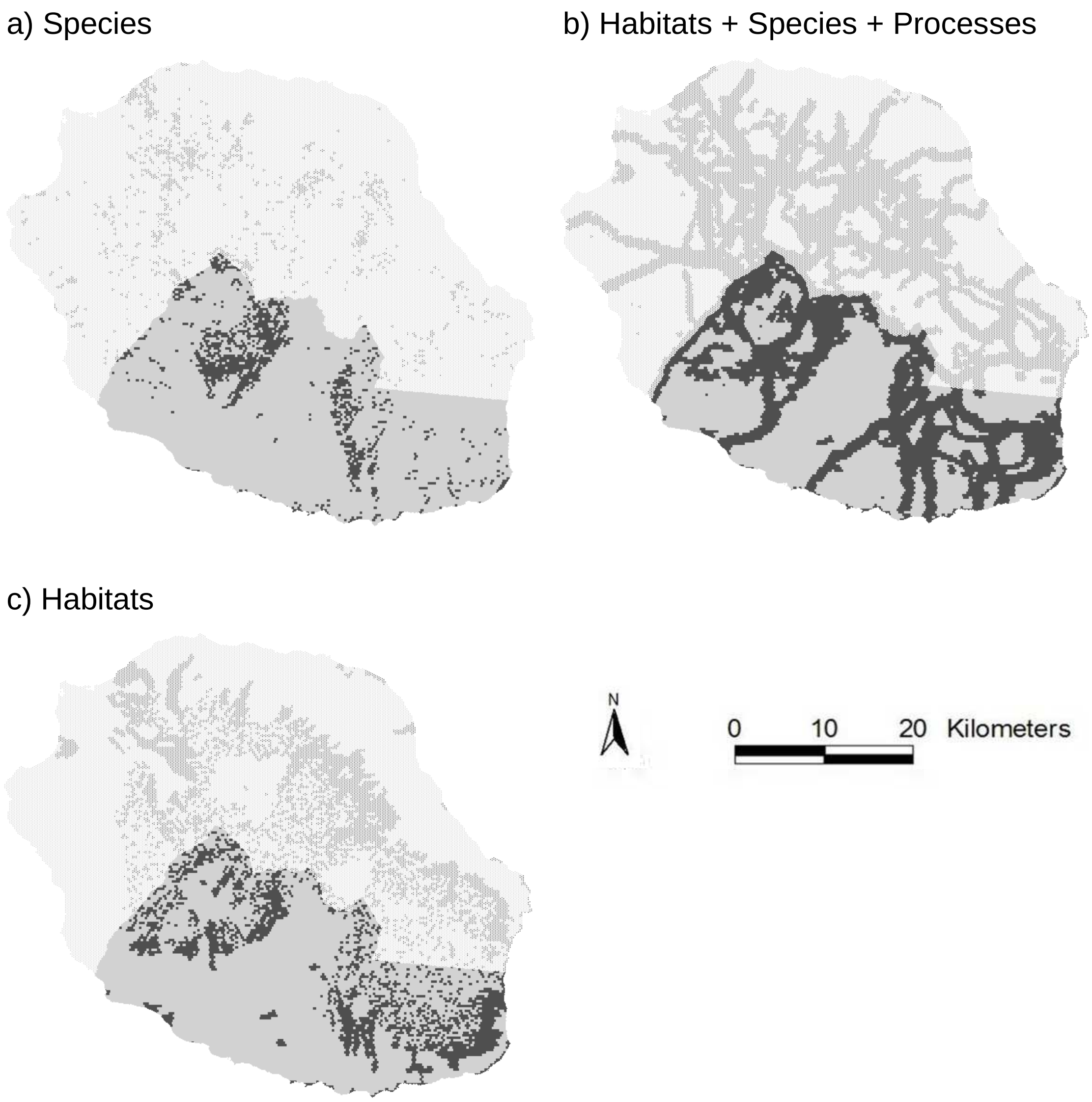

Fig. 4 – Maps of the regional near-minimum sets that achieved the lowest (a) and the highest (b) incidental representation of local-scale biodiversity surrogates and of the regional near-minimum set that achieved the best compromise between incidental representation, number of regional-scale surrogates used for the area-selection and total area selected (c). The planning units selected in each near-minimum set are shown in darker grey. The area outside the local planning domain is shaded.

performed comparative levels of incidental target achievement (68% of local-scale species and 67% of local-scale habitats) but for only half as much area (~ 220 sq km, Fig. 3, see also Fig. 4). Of all the near-minimum sets, it is the one that achieved the best compromise between total area selected and incidental target achievement.

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3.1.2. Comparison to the random selections

The average area selected by the three groups distinguished on the x axis of Fig. 3 (see also Table 2) was used to fix the area to be selected by three random selections (R1, R2 and R3 in order of increasing area). Differences in target achievement between the random selections and their corresponding near-minimum sets were higher (up to 26% of features between HS and R2) when near-minimum set combinations targeted regional-scale habitats (i.e. H, HS, HP and HSP; Fig. 3). Selections targeting processes (P) or species (S) performed rather similarly to random selections (R3 and R1, respectively).

3.1.3. Features representation

The regional near-minimum set targeting species (S) achieved targets for three species and two habitats. In the case of all the other regional near-minimum sets, the incidental target achievement of local-scale biodiversity surrogates followed three trends: features meeting their targets under all circumstances, features never meeting their targets and features meeting their targets under certain circumstances. This was generally correlated to the amplitude of the conservation targets.

The features that always met their targets incidentally were ten of the 11 local-scale species with targets ≤ 30% and nine of the 11 local-scale habitats with targets ≤ 45 % (Table 1). In general, they also met their targets in the random selections R2 and R3. In contrast, local-scale biodiversity surrogates with conservation targets ≥ 84% of their current extent never met their targets.

The three local-scale coastal species (*Delosperma napiforme*, *Chamaesyce viridula* and *Pemphis acidula*), also mapped and targeted at 100% at regional scale, did not meet their targets in regional near-minimum sets targeting species. When compared, the CBNM point locality records used at regional scale and the distribution ranges obtained by expert mapping at local-scale overlapped only partly.

3.2. The irreplaceability approach

The spatial correlation between patterns of irreplaceability values of both scales was positive and varied from low (Spearman’s rank correlation coefficient: 0.34) to very high (0.94; Table 4). Again, regional-scale species were the least effective biodiversity surrogates, since their irreplaceability values were always the least correlated with the irreplaceability values of all local-scale surrogates (Table 4; see also Fig. 6b). On the contrary, the irreplaceability values of regional-scale habitats and processes (HP) were, on average, the

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Table 4 – Spearman's rank correlation coefficients between the regional-scale and the local-scale irreplaceability values. Bold values indicate remarkable low and high coefficients. The lowest coefficients are observed for regional- and local-scale summed solutions targeting species only (S); the maximum coefficient is observed between regional- and local-scale summed solutions for habitats only (H); the highest coefficient with local-scale summed solutions targeting species (S) is observed with regional-scale summed solutions (H); the regional-scale summed solution targeting habitats and processes (HP) is the one that is overall the best correlated with the local-scale summed solutions.

		Regional						
		H	**S**	**P**	**HS**	**HP**	**SP**	**HSP**
Local	**H**	**0.94**	**0.36**	0.49	0.88	0.57	0.49	0.57
	S	**0.63**	**0.34**	**0.40**	0.60	**0.41**	**0.40**	**0.40**
	P	0.45	**0.35**	0.80	0.47	0.76	0.79	0.76
	HS	0.91	**0.35**	0.47	0.85	0.56	0.47	0.55
	HP	0.55	**0.37**	0.77	0.57	**0.81**	0.76	0.80
	SP	0.46	**0.35**	0.79	0.48	**0.76**	0.78	0.76
	HSP	0.55	**0.37**	0.76	0.57	**0.80**	0.76	0.80

most correlated with the irreplaceability values of all local-scale surrogates; but the highest coefficient between irreplaceability values of both scales was found between habitats (Table 4 and Fig. 6a). The correlation of irreplaceability values of local-scale species was generally low with the irreplaceability values of regional-scale biodiversity surrogates (Fig. 6b), but was the highest with irreplaceability values of regional-scale habitats (Table 4).

4. Discussion

Careful attention is needed to understand where fine-scale data will lead to marginally better conservation decisions (Conroy and Noon, 1996). This study indicated that regional assessments were relatively effective in achieving conservation targets for local-scale biodiversity surrogates, but that results varied depending on the type of regional- and local-scale biodiversity surrogates. Area requirement also varied greatly between near-minimum sets of regional-scale biodiversity surrogates. In general, the larger tracts of land was selected,

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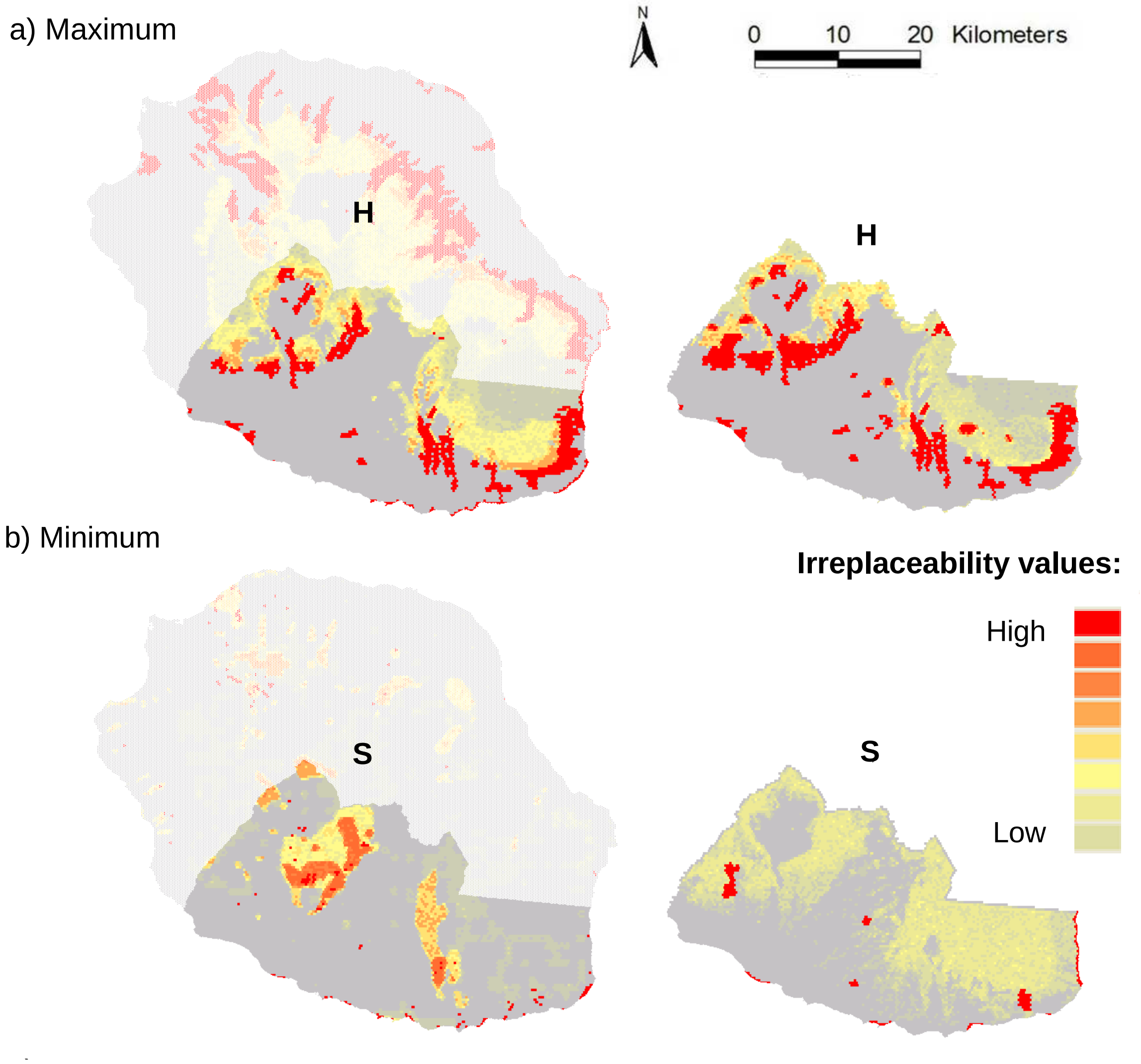

Fig. 5 – Maps of the irreplaceability values for the maximum (a) and the minimum (b) Spearman rank correlation coefficients when the regional (the area outside the local planning domain is shaded) and local Marxan summed solutions were compared pair-wise.

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the higher the incidental representation of local-scale biodiversity surrogates was (Fig. 3), which illustrated the trade-off between level of representation and amount of land required (Reyers et al., 2002; Wessels et al., 1999).

4.1. Regional-scale species as biodiversity surrogates

The issue of the effectiveness of species as biodiversity surrogates, in particular as cross-taxon indicators, has often been addressed in the conservation literature and is a complex one (see Reyers and van Jaarsveld, 2000; Reyers et al., 2000; Warman et al., 2004a).This study

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confirms that targeting species at a regional scale proved to be a relatively ineffective surrogacy option for the incidental representation of most of the local-scale biodiversity surrogates (see also Reyers et al., 2000; Warman et al., 2004b). The low spatial correlation between their irreplaceability values and those of all local-scale biodiversity surrogates further indicates that they stand little or no chance to incidentally represent local-scale biodiversity surrogates at target level. The fact that the area requirement of their near-minimum set was inferior to the area requirement of any of the near-minimum sets targeting local-scale biodiversity surrogates is problematic for a regional-scale biodiversity surrogate (Reyers et al., 2000).

4.2. Regional-scale habitats as biodiversity surrogates

The best trade-off between a maximum representation and least area selected was obtained with the selection based on regional-scale habitats. The high similarity between patterns of irreplaceability values of habitats at both scales might have been positively affected by the nestedness of the classifications. Even if this aspect might be considered as a particular case, it nonetheless demonstrated that the coarser classification (i.e. used at regional scale) was a useful regional-scale surrogate in Réunion. Habitats (or land types) are expedient biodiversity surrogates because they are relatively easy and inexpensive to derive and revise (Ferrier, 2002; Pressey, 2004) so they can be mapped over large areas. They are often defined by generalisations about environmental variation so are assumed to match the distribution of at least a proportion of species in a given region (Oliver et al., 2004; Pressey, 2004; Stoms et al., 2005). As such, they usually insure a good incidental representation for at least wide-spread species (e.g. Wessels et al., 1999; Reyers et al., 2002; Lombard et al., 2003; Oliver et al., 2004; Stoms et al., 2005) as was also the case here.

4.3. Regional-scale processes as biodiversity surrogates

The findings clearly highlight that planning for patterns and processes considerably increases area requirement compared to planning for patterns only. Close to half the area of the local planning domain was necessary to represent processes in conservation networks. This, at the same time, insured a high incidental representation of local-scale species and habitats at target level, but overall proved a rather ineffective surrogacy option since its results were similar to randomly selecting planning units for an equivalent area. However, the integration of processes in an area-prioritisation for both patterns and processes can be done in a more

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efficient manner than was the case here. Here, all targets were fixed at 100% of extant areas, but this might not be necessary for all processes. Besides, the configuration of some processes are spatially-flexible (e.g. upland-lowland gradients) (Rouget et al., 2003) and when treated as such, their overlap with the distribution of important biodiversity patterns can be maximised. Finally, the area-selection protocol can be adapted in order to enhance the efficiency of the final conservation network. For instance, Cowling et al. (2003) designed a protocol in seven stages, where spatially-fixed processes were integrated at stage two and two spatially-flexible processes were integrated at stages five and six, respectively.

4.4. The incidental representation of local-scale biodiversity surrogates at target level

The discrepancy observed between the distributions of three threatened coastal plant species analysed in this study highlights the difficulty of obtaining consistent distributions of imperilled species, and reliable species data in general (Reyers et al., 2001). The datasets used at both scales were indeed obtained from reliable sources (a botanical institute and a field technician with deep knowledge on the flora of this part of the island). It is likely that this impediment is inherent to many surrogacy analyses and conservation assessments based on species data, but that it was unmasked here because two different datasets were used for the same taxa. This underpins the need of good field survey efforts for the mapping of threatened species.

None of the processes identified for this assessment ever met their targets incidentally. This is the reason why they need to be explicitly tackled in conservation assessments (Pressey et al., 2003).While small-scale ecological and evolutionary processes can be captured when planning for the representation of biodiversity patterns (Rouget et al., 2003), the same does not apply to those processes that operate over large areas and in particular spatial configurations (Fairbanks and Benn, 2000; Desmet et al., 2002; Moritz, 2002; Noss, 2003). These processes usually are supported by geographic and environmental interfaces and gradients, and migratory corridors (e.g. Cowling and Pressey, 2001) and insure the persistence of functional ecosystems, but also resilience to rapidly-occurring climate change (IPCC, 2007) by providing migratory pathways to new ecological niches (Hunter et al., 1988; Noss, 2001).

Local-scale habitats and species with high percentage targets were typically those not reaching their targets incidentally in near-minimum sets. High targets usually concern threatened and/or rare species and fragmented and highly transformed habitats, the features generally in most urgent need of conservation. Rare and threatened biodiversity features have

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recurrently been missed in area-selections for other biodiversity surrogates (Reyers et al., 2000; Reyers et al., 2002; Lombard et al., 2003; Rouget, 2003; Warman et al., 2004a; Stoms et al., 2005). Threatened species are usually correlated to habitat transformation (i.e. loss, degradation and fragmentation; Wilcove et al., 1998; IUCN, 2004; Ricketts et al., 2005). In Réunion Island, threatened species and degraded habitats were mostly in lowland areas, where most of the urban and agricultural activities are concentrated. The distribution of these imperilled biodiversity features could be conveniently predicted from accurate and fine-scale land cover information (Chapter 4). Other threatened species (in particular birds) were restricted to less accessible parts of the landscape, where they were also associated to more cryptic threatening processes such as predation from introduced mammals (Probst et al., 2000). The involvement of expert knowledge and the use of lists of threatened taxa should complement the use of fine-scale land cover data.

4.5. Concluding remarks: a spatial strategy to the choice of biodiversity surrogates in conservation assessments

The findings suggest that a spatial strategy based on a complementary set of regional- and local-scale biodiversity surrogates can be an effective approach to conservation assessments. All three types of biodiversity surrogates were shown necessary for maximum biodiversity representation and persistence (Noss, 1990; Fairbanks and Benn, 2000; Margules and Pressey, 2000), but in a manner characteristic of the coarse filter/fine filter approach. Coarse filters are habitat-type surrogates and spatial components for processes that typically insure the representation and persistence of widespread or cryptic species and ecosystem processes (Hunter et al., 1988; Noss, 1996; Noss, 2002; Stoms et al., 2005), while fine filters usually are rare or imperilled species, assemblages and communities that are normally missed by coarse filters (Noss, 2002; Stoms et al., 2005). Noss (2003) provides a good checklist to help identify these features. The results further highlighted that a spatial strategy consisting of targeting the coarse filters (processes included) at regional scale and the fine filters at local scale, may be appropriate. Fine-scale mapping and surveying effort for fine filters can be principally focused on transformed areas (Rouget, 2003). Such a generalised framework could also be adapted with the incorporation of surrogates for wide-ranging species at regional scale where this is relevant (Poiani et al., 2000).

Fine filters usually are features under the most urgent need for conservation (Stoms et al., 2005). If opportunities to safeguard them arise, they should not be postponed until a regional assessment is accomplished. Thus, a foreseeable advantage of basing regional- and

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local-scale assessments on complementary coarse and fine filters (Noss, 2002), is that undertaking these assessments independently in time should not impair the efficiency of the final conservation network significantly. Providing that assessments over larger areas are likely to increase the efficiency of conservation networks (Erasmus et al., 1999), spatial frameworks integrating regional- and local-scale assessments are likely to be more the rule than the exception (e.g. The Nature Conservancy, www.nature.org). The integrated approach proposed here needs to be more elaborated and further investigated.

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Appendix A. Species and habitats used as biodiversity surrogates at regional and local scale

A.1. Habitats at regional scale (Strasberg et al., 2005)

Recent lava flows, Subalpine macrohabitat, Mountain macrohabitat, Submountain macrohabitat, Lowland macrohabitat and Wetlands.

A.2. Habitats at local scale (Strasberg et al., 2005)

Semi-dry forest, Lowland rainforest, Wetlands, Leeward submountain rainforest, Subalpine sophora thicket, Tamarind forest, Leeward mountain rainforest, Winward submountain rainforest, Philippa mountain thicket, Submountain mesic forest, Winward mountain rainforest, Subalpine heathland, Coastal habitats, Pandanus mountain humid thicket, Pandanus humid thicket, Pioneer vegetation on lava, Subalpine wet grasslands and Subalpine shrubland on lapili.

A.3. Species at regional scale (animals and plants)

A.3.1. Animals (SEOR, Nature et Patrimoine)

Bat : *Mormopterus acetabulosus*

Terrestrial birds: *Collocalia francica, Phedina borbonica, Pseudobulweria aterrima, Pterodroma baraui, Circus maillardi, Coracina newtoni, Hypsipetes borbonica, Saxicola tectes, Terpsiphone bourbonnensis* and *Zosterops olivaceae*.

Marine birds: *Puffinus lherminieri bailloni, Phaethon lepturus* and *Puffinus pacificus* assemblage.

Reptiles: *Phelsuma borbonica* and *Phelsuma inexpectata*.

A.3.2. Plants (CBNM)

Carissa spinarum, Delosperma napiforme, Dombeya populnea, Gastonia cutispongia, Chamaesyce viridula, Hernandia mascarenensis, Obetia ficifolia & *Pemphis acidula*

A.3. Species at local scale (plants) (E. Rivière)

Cossinia pinnata, Eugenia buxifolia, Securinega durissima, Sophora denudata, Claoxylon sp and *Dombeya* sp assemblage; *Tournefortia argentea; Pemphis acidula; Delosperma napiforme; Ochrosia b.* & *Sideroxylon m.; Chamaesyce viridula; Chamaesyce goliana; Psiadia retusa; Erica* sp, *Agauria buxifolia, Phylica nitida, Hubertia tomentosa*, and *Stoebe passerinoides* assemblage; *Weinmania tinctoria, Monimia rotundifolia, Nuxia verticillata, Chassalia gaertneroides, Gaertnera vaginata, Hypericum lanceolatum, Erica arborescens* and *Dombeya* sp assemblage; *Cossinia pinnata, Eugenia buxifolia, Securinega durissima, Sophora denudata* and *Claoxylon* sp assemblage; *Cossinia pinnata, Eugenia buxifolia* and *Securinega durissima* assemblage*; Sophora denudate; Scaevola taccada; Erica reunionensis; Cyathea glauca* & *C. excelsa; Labourdonnaisia calophylloides, Agauria saliciffolia* and *Mimusops maxima* assemblage*; Claoxylon glandulosum;* and *Acacia heterophylla*.

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Chapter 4

Mapping threatened ecosystems: the relative importance of scale, ecosystem classification and habitat transformation

A B S T R A C T

Biodiversity assessments identify biodiversity features under greatest risk of extinction, in order to promote their conservation. In recent years, they have been applied more widely at ecosystem-level. In South Africa, the national legislation provide for the listing of threatened ecosystems. Various criteria such as extent and rate of habitat loss, species diversity and habitat fragmentation can be used to identify threatened ecosystems; the habitat loss criterion was investigated here. Three ecosystem classifications and three land cover databases of the Little Karoo region, South Africa, were used in order to explore the sensitivity of the assessment to the use of these different databases. Habitat loss was quantified on land cover information. The quantification of habitat loss varied across land cover databases due to different levels of mapping accuracy, and reflected the spatial patterns of threatened ecosystems of all three ecosystem classifications. Less than 14% of extant areas were classified threatened with the coarsest land cover maps, in comparison to 30% with the finest one; and less than 9% of ecosystem types were threatened with the coarsest land cover maps, but between 15 and 23% were threatened with the finest one. Results revealed that the identification of threatened ecosystems is more sensitive to the accuracy of habitat loss quantification than the resolution of ecosystem classifications. Under budgetary constraints, priority should be given to generating more detailed land cover maps than more detailed ecosystem maps for the assessment of threatened ecosystems.

Keywords: Conservation status; Conservation targets; Habitat loss; Systematic conservation planning

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1. Introduction

Biodiversity assessments highlight the biodiversity features threatened with extinction and promote their conservation (IUCN, 2001). Efforts in risk assessments have primarily been focused on the species level of biodiversity (Noss and Peters, 1995; Bonn and Gaston, 2005) with governmental agencies and other non-governmental organisations producing threatened species lists (Possingham et al., 2002). For instance, the IUCN Red List of Threatened Species is the most comprehensive database detailing the conservation status of plant and animal taxa at global scale (Rodrigues et al., 2006). Its criteria have been used extensively in official national listing efforts (Miller et al., 2007). For over three decades, the World Conservation Union (IUCN) has catalogued species and subspecies in the threatened categories of critically endangered, endangered and vulnerable, through a set of quantitative criteria that takes into consideration the decline, distribution, rarity and fluctuations of populations (IUCN, 2001).

Risk assessments at ecosystem level have the potential to overcome some of the biases linked to species data, such as the taxonomic bias towards vertebrate species and vascular plants and spatial sampling biases (Maddock and Du Plessis, 1999; Pressey, 2004). They also directly address the primary cause of biodiversity decline, i.e. habitat destruction (Orians, 1993; Noss et al., 1995). Consideration of biodiversity at ecosystem level also presents the advantage to encompass some of the processes that sustain biodiversity persistence over time (Noss, 1996). Hence, in recent years, there has been more consideration of ecosystem-level approaches (Reyers et al., 2001). Noss et al. (1995) and Noss and Peters (1995) assessed threatened "ecosystems" (i.e. vegetation, natural communities or habitat types) of the United States and Olson and Dinerstein (1998, 2002) and Burgess et al. (2004), threatened "ecoregions", at global and continental scales, respectively. They used scoring methods based on multiple criteria including habitat loss, degradation and fragmentation which were adapted from the IUCN Red List approach. In other instances, ecosystems have been listed based on qualitative (e.g. European Union's Habitats Directive, http://ec.europa.eu) or similar quantitative criteria (e.g. in Australia, www.awc.org.au), but attempts are overall rather few (T. Smith, unpublished work).

In South Africa, Pierce et al. (2005) classified threatened ecosystems of the Subtropical Thicket biome using vegetation types as ecosystems. The assessment consisted of quantifying habitat loss of each vegetation type, and classifying it into a conservation status category according to the difference between its conservation target and extant area, both expressed as a percentage of the original (i.e. pre-transformation) extent. Habitat loss was

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quantified on a land cover map. The categorisation of endangerment aimed to "provide land-use decision makers with information enabling them to make decisions that would enhance instead of compromise the achievement of biodiversity targets" (Pierce et al., 2005). The assessment was performed with a systematic conservation planning basis, a scientific approach adopted by South Africa since the 1990s (Knight et al., 2006). Systematic conservation planning uses accountable methods to plan for the conservation of a representative sample of all biodiversity and to maintain its persistence over time (Margules and Pressey, 2000). It is data- and target-driven (von Hase et al., 2003). Biodiversity, or conservation, targets (i.e. the respective minimum representation of biodiversity features to be held in a conservation network (Pressey et al., 2003)) provide an explicit and defensible reference point for conservation decisions (Margules and Pressey, 2000). Targets used by Pierce et al. (2005) were derived from species-area curves. They corresponded to estimates of the fraction of area required, to obtain an agreed upon conservation goal, of representing one occurrence of 75% of the component species of each vegetation type (Desmet and Cowling, 2004). The robustness of this approach lies in the use of explicit and defensible conservation targets.

The South African legislation provides for the listing of threatened ecosystems through its National Environmental Management: Biodiversity Act, 2004 (No. 10 of 2004; DEAT, 2004). Three categories of threatened ecosystems are denoted by the Act (i.e. critically endangered, endangered and vulnerable). The Act, however, does not specify how threatened ecosystems should be identified, hence, the South African National Biodiversity Institute (SANBI) is currently developing the criteria and thresholds for the categorisation. Five criteria have been proposed: irreversible habitat loss, rate of habitat loss, habitat fragmentation, limited extent and imminent threat, and number of threatened species per ecosystem. The criterion of irreversible habitat loss is ready for use. And has already been tested by Pierce et al. (2005) (as described above) and in the first South African National Spatial Biodiversity Assessment (NSBA; Driver et al., 2005). SANBI defines irreversible habitat loss (hereafter habitat loss) as habitats that have undergone complete transformation (e.g. urban areas) or severe degradation (e.g. areas with high density of invasive species). The thresholds between threatened categories were last revised at an expert workshop, in December 2006. They are still based on rationales of biodiversity persistence like Pierce et al. (2005), but they remain somewhat arbitrary due to limitations in scientific knowledge. This study focuses on this criterion only.

The assessment of threatened ecosystems as proposed by SANBI is potentially sensitive to the data used, the criteria and the thresholds. This study looks at testing the role of the data

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used (i.e. an ecosystem map and a land cover map). There is no standardised approach to ecosystem and land cover mapping. Ecosystem types have been derived at various spatial scales, on varying classification systems (Orians, 1993; Noss, 1996), while, similarly, land cover data are produced on varying methodologies that result in databases of various accuracies (Reyers et al., 2001; Rouget et al., 2003). The Little Karoo region is a global biodiversity hotspot (i.e. the Succulent Karoo) in south-western South Africa. This study site provided the materials to explore the effect of both types of data on conservation status assessments. Three ecosystem maps and three land cover maps obtained from independent sources overlapped in the region. The threatened status of the ecosystem types was determined based on the latest thresholds from SANBI and the nine combinations of ecosystem and land cover maps. The threatened ecosystems and their spatial patterns were compared and the results were interpreted in terms of the relative benefits of land cover or ecosystem mapping.

2. Material and methods

2.1. Study area

The Little Karoo, or Klein Karoo, is a region of the Western Cape Province in south-western South Africa (Fig. 1). The region forms a basin of approximately 19 341 square kilometres in

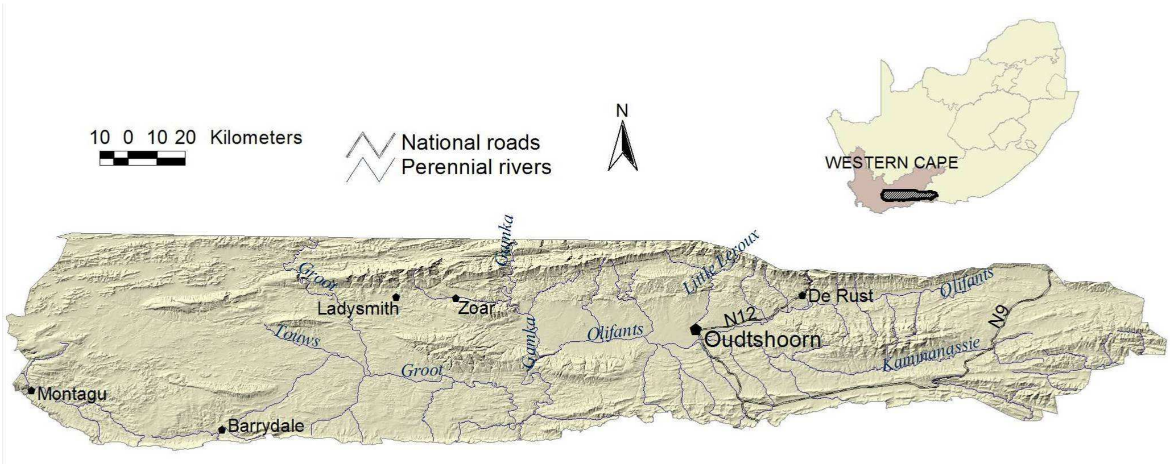

Fig. 1 – Map of the Little Karoo region showing the topography as a background and the main river system, the national roads and the main towns. The location of the region in South Africa is indicated on the smaller map on the top right-hand corner.

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the southeast of the Succulent Karoo biome and is one of the twelve bioregions of the biome (in Lombard et al., 1999). The Succulent Karoo biome qualifies as one of only two global biodiversity hotspots that are entirely semi-arid (the other is the Horn of Africa; www.biodiversityhotspots.org) and is home to the world's richest succulent flora (Cowling et al., 1998). Using a Red Data Book plant species dataset to assess conservation priorities across the biome, Lombard et al. (1999) found that the Little Karoo is the bioregion with the highest level of endemicity and, second to the Gariep Centre bioregion, in the most in need of conservation action based on vulnerability and endemicity. The most extensive pressure on biodiversity is livestock grazing, goat, sheep, ostrich and small game ranching being its dominant land uses, and signs of overgrazing, in particular from ostrich farming, are evident over much of the landscape (Thompson et al., 2005). Based on the recent land transformation assessment of Thompson et al. (2005), approximately 25% of the region is considered as severely transformed, 37% as moderately transformed (with potential for restoration) and 38% as pristine. The region offers diverse tourist attractions with the famous Garden Route, the Klein Karoo Wine Route, seasonal wild-flower watching and hiking trails. Human population is sparse and the main town (Oudtshoorn) contained approximately 85 000 inhabitants in 2001 and has low population growth (www.dwaf.gov.za).

2.2. Mapping terrestrial ecosystems

Terrestrial ecosystems in the Little Karoo were delineated on existing vegetation and habitat maps (Table 1). They were derived on different classification systems that produced ecosystem types at different resolutions (Fig. 2). These were Broad Habitat Units (BHUs; Cowling and Heijnis, 2001), South African National Biodiversity Institute (SANBI) vegetation units for South Africa, Lesotho and Swaziland (Mucina et al., 2005) and vegetation units of the Little Karoo region mapped by Vlok et al. (2005). Each map provides the estimated original extent and distribution of their respective ecosystem components.

2.2.1. Broad Habitat Units

BHUs were mapped as biodiversity surrogates for the Cape Action Plan for the Environment (CAPE) conservation plan of the Cape Floristic Region (CFR; Cowling and Heijnis, 2001). The map is for use at a scale of 1:250 000 and coarser. Cowling and Heijnis (2001) mapped 88 BHUs across the CFR by intersecting information on climate, geology and topography and used vegetation types (Low and Rebelo, 1996) to guide the classification under certain circumstances, as well as expert opinion. The physical data used are well-established

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Table 1 – Ecosystem databases analysed for the assessment of the conservation status of their respective ecosystem types and conservation targets in percent of original extent (see section 2.4 for information on the setting of targets)

Ecosystem types	Authors	Scale	Conservation targets
26 Broad Habitat Units (BHUs)	Cowling and Heijnis (2001)	1:250 000	10, 15 and 25%
40 SANBI vegetation units	Mucina et al. (2005)	1:250 000	16 to 34%
369 Little Karoo vegetation units	Vlok et al. (2005)	1: 50 000	16 to 34%

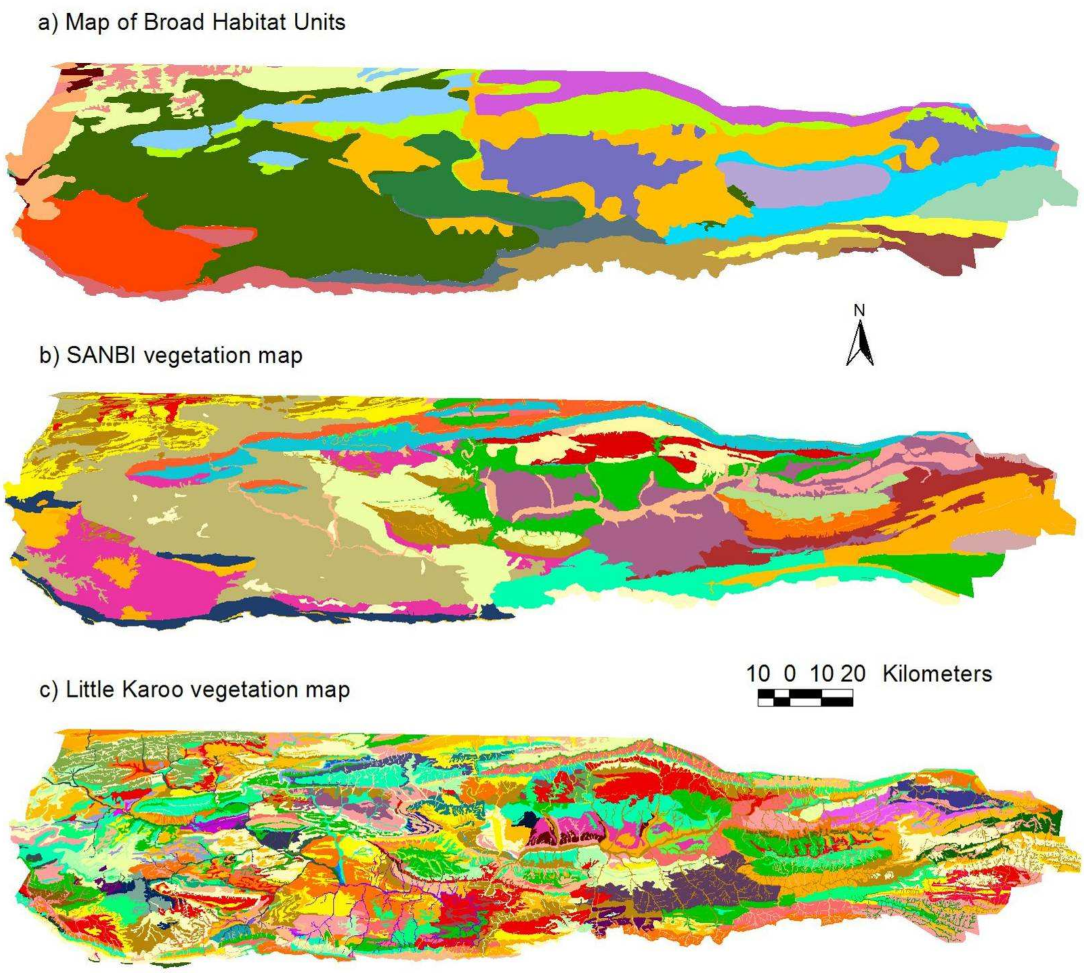

Fig. 2 – Vegetation maps used for the conservation status analysis (see Table 1 for details).

correlates of plant species and vegetation patterns in the CFR (Cowling and Heijnis, 2001), so the authors are confident that BHUs are good predictors of plant diversity in the region.

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The original map of BHUs was clipped to the extent of the Little Karoo region in ArcView 3.2 (ESRI, Redlands, CA, USA) and a new map of 26 BHUs represented in 95 polygons was obtained (Fig. 2a). Each BHU ranges from 17 to 444 049 ha, represented in one to eleven polygons with patch size from <1 to 429 858 ha.

2.2.2. SANBI vegetation units

SANBI vegetation units were derived and mapped by Mucina et al. (2005), to provide floristic-based vegetation units for South Africa, Lesotho and Swaziland. The map was derived on plant species distributions, existing vegetation maps and peer review from various experts in different parts of the country. The database is of use at a scale of 1:250 000 and coarser. It represents 441 vegetation units across all three countries.

The original map was clipped (ArcView 3.2, ESRI, Redlands, CA, USA) to the extent of the Little Karoo region to produce a new map that consisted of 40 vegetation units represented in 608 polygons (Fig. 2b). Each vegetation unit ranges from 17 to 372 795 ha, represented in one to 130 polygons with patch size from <1 to 339 509 ha.

2.2.3. Little Karoo vegetation units

Vlok et al. (2005) derived a map of the vegetation of the Little Karoo region to inform sustainable land-use decision making at fine-scale (Fig. 2c). The map is of use at a scale of 1:50 000 and coarser. Vegetation units were identified and mapped through systematic, 6-month intensive field survey, on the basis of the unique ecological characteristics of their habitat and structural and floristic composition of their dominant species community. The map consists of 369 vegetation units digitised in 2 728 polygons. Each vegetation unit ranges from 48 to 48 454 ha, represented in one to 83 polygons with patch size from six to 48 454 ha.

2.3. Mapping habitat transformation

Three land transformation databases were available to quantify habitat loss in every ecosystem (Table 2). These were the CAPE land cover map (Cowling et al., 1999; Rouget et al., 2003), the National Land Cover (NLC) map of 1996 (Fairbanks et al., 2000) and the transformation map of the Little Karoo (Thompson et al., 2005). Where needed, the original land cover classes were systematically reclassified in a binary classification of transformed or natural for the three land cover databases. The transformed class incorporated transformation by urbanisation, cultivation (including forestry plantations), severe overgrazing impacts and other severely-impacted unsustainable land use practices (e.g. mining) or high-density stands

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of invasive alien plant species. These areas had undergone complete or acute transformation to the degree that species structure and ecosystem functions were totally lost or severely degraded, respectively. These habitats retained no potential to restore naturally if the threatening factor is removed, contrarily to moderately degraded areas (e.g. low alien invasion) (Thompson et al., 2005). Hence, all transformed habitats were here considered irreversibly lost, i.e. retaining no natural or long-term viable biodiversity content.

The quantification of habitat loss within ecosystem types is a simple procedure that consists of overlaying a map of ecosystems and a land cover map into a geographic information system (GIS) (Reyers et al., 2001). The three land cover maps were intersected with each map of ecosystem types in ArcView 3.2 (ESRI, Redlands, CA, USA), in WGS84 UTM/Zone 34S. This produced nine assessments. For each of them, habitat loss was quantified by summarising the extent of lands classified as transformed within each ecosystem type. These estimates were then compared between the three assessments of a same ecosystem classification (Friedman tests, R Development Core Team, 2007). I also tested if the differences of estimates were consistent across land cover databases (i.e. if high estimates with one land cover are correlated with high estimates with another one; Spearman’s rank correlation coefficients, R Development Core Team, 2007).

2.3.1. CAPE land cover map

The CAPE land cover database was generated as part of the CAPE project for the conservation plan of the CFR (Cowling et al., 2003, Rouget et al., 2003) from single-date Landsat Thematic Mapper imagery captured between December 1997 and February 1998 and for use at 1:250 000 scale mapping applications (Lloyd et al., 1999). The land classification was primarily produced from an unsupervised (automatic) classification used to produce clusters of pixels with similar spectral characteristics and secondarily by manual recoding of misclassifications based on expert knowledge and interpretation of ancillary datasets.

A version of the original land cover map, called CAPE untransformed areas, had been produced in order to provide a map of remnants of natural habitats only, to the CAPE project (Cowling et al., 1999). These remnants corresponded to the natural land cover classes with medium and none to low alien invasion. All areas transformed by urbanisation, cultivation (including heavily grazed), plantations or high-density aliens had been classified as transformed. This version was used for this analysis, without further alteration (Fig. 3a).

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Table 2 – Land cover databases used for the quantification of habitat loss of the three assessments of the conservation status of ecosystem types of each ecosystem classification (Table 1)

Land cover maps	Authors	Mapping unit	Scale	Data capture period
CAPE	Cowling et al. (1999)	25 ha	1:250 000	1997-1998
National (NLC)	Fairbanks et al. (2000)	25 ha	1:250 000	1994-1996
Little Karoo	Thomspon et al. (2005)	20-25 ha	1:100 000	2000-2003

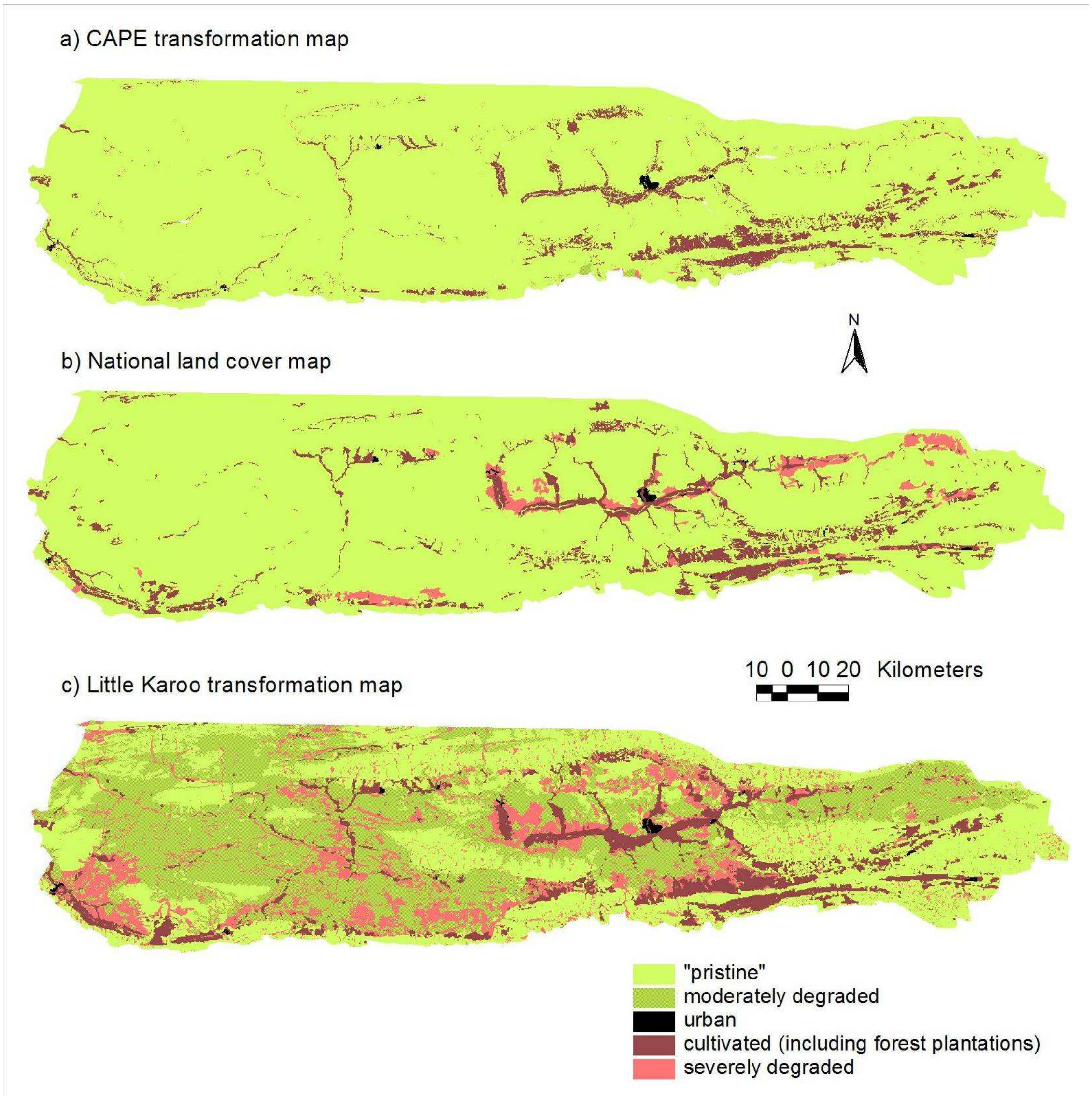

Fig. 3 – Land cover maps used for the conservation status assessments (see Table 2 for details).

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2.3.2. The national land cover

The NLC data (Fairbanks et al., 2000) is a land-cover database for South Africa, Lesotho and Swaziland that was derived from the interpretation of single-date Landsat Thematic Mapper imagery captured from 1994 to 1996 and designed for 1:250 000 scale mapping applications (Fairbanks et al., 2000). It was released in 2000 by the Council for Scientific and Industrial Research (CSIR) and Agricultural Research Council (ARC). The database counts 31 land-cover classes that were mapped using manual photo-interpretation.

Areas considered transformed for this analysis were severely degraded lands due to excessive fuel-wood removal, over-grazing and subsequent soil erosion, or areas under crop cultivation, forestry plantations, urbanised areas and mines/quarries (Fig. 3b) (see Reyers et al., 2001). Invasive alien species stands had not been mapped in this database.

2.3.3. The Little Karoo transformation map

The Little Karoo land cover (Thompson et al., 2005) was produced in order to provide a spatial quantification of grazing impact and land uses for the Little Karoo region to be suitable for mapping application of 1:100 000 scale and coarser. The land cover classes were derived on a combination of multi-seasonal Landsat and MODIS NDVI satellite imagery captured within the overall period 2000-2003. The land classification was based on degradation models calibrated to each biome or habitat from the vegetation map of Vlok et al.

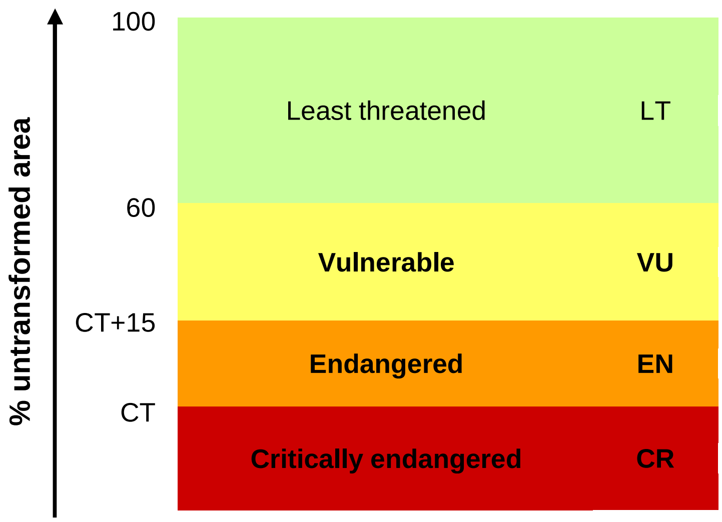

Fig. 4 – Classification of ecosystems into four conservation status categories, the threatened categories appearing in bold, based on % of untransformed area relative to the original extent, and the conservation target (CT; adapted from Rouget et al., 2004).

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(2005) and using the vegetation spectral index (i.e. NDVI, normalised difference vegetation index). The final database provides the degradation status of the Little Karoo vegetation in three levels of transformation (i.e. “pristine”, “moderate” and “severe”) due to grazing impacts, alien plant invasion and total vegetation loss as a result of cultivation, settlement development and/or water resource retention land uses.

For this analysis, all “pristine” and “moderate” transformation classes, which, respectively, correspond to essentially untransformed sites and sites where alterations of plant community composition and ecosystem functions by domestic livestock are low and essentially naturally reversible, were reclassified as natural. Areas falling in the “severe” class, i.e. corresponding to sites with plant communities and ecosystem functions that have been severely altered by domestic livestock, or cultivated, urbanised and heavily alien invaded areas, were reclassified as transformed (Fig. 3c).

2.4. Mapping threatened ecosystems

Conservation targets were available for all ecosystem types from existing conservation planning initiatives (Table 1). Each target corresponded to a percentage of original (i.e. pre-transformation) extent of each ecosystem type. The targets for BHUs had been defined in CAPE conservation plan with consideration of the level of biological heterogeneity (beta and gamma diversity) of individual BHUs (Pressey et al., 2003). They took the values 10, 15 or 25% of original extent. The targets of SANBI and the Little Karoo vegetation units had both been defined on the methods by Desmet and Cowling (2004) for the NSBA 2004 and for the fine-scale planning of the Little Karoo region underway, respectively. NSBA targets of SANBI vegetation units ranged from 16 to 34% of original extent; the targets for the Little Karoo vegetation units also ranged from 16 to 34% of original extent.

The thresholds between conservation status categories, fixed by a workshop of experts in December 2006, are presented in Fig. 4 (M. Rouget, personal communication). The category critically endangered (CR) was assigned to ecosystem types where the area targeted for conservation (i.e. conservation target x original area) > extant area; where the difference between the area targeted for conservation and extant area < 15% of original extent, the ecosystem type was categorised endangered (EN); where extant area < 60% of original extent, the ecosystem type was classified vulnerable (VU); and where extant area ≥ 60% of original extent, the ecosystem type was categorised least threatened (LT).

The nine assessments of habitat loss were used to produce corresponding assessments of threatened ecosystems. The spatial patterns of conservation status were mapped in ArcView

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Table 3 – Extent of habitat loss relative to the original area of each ecosystem type (i.e. BHUs, SANBI vegetation units and Little Karoo vegetation units) based on its quantification from the three land cover databases: the CAPE land cover map (CAPE), the NLC and the Little Karoo transformation map (LK). (Min. = minimum; Med. = median; and Max. = maximum)

	BHUs (%)			SANBI vegetation units (%)			Little Karoo vegetation units (%)		
	CAPE	NLC	LK	CAPE	NLC	LK	CAPE	NLC	LK
Min.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Med.	1.94	3.53	11.22	0.40	1.47	14.56	1.98	3.43	13.33
Max.	41.96	51.11	55.90	69.36	75.75	88.08	76.78	97.32	96.15

Table 4 – Spearman's rank correlation coefficients between habitat loss estimates within ecosystem types based on the three land cover databases: the CAPE land cover map (CAPE), the NLC and the Little Karoo transformation map (LK) (Kruskal-Wallis test between columns, p-value = 0.06081)

	BHUs	SANBI vegetation units	Little Karoo vegetation units
CAPE/NLC	0.954	0.946	0.8
NLC/LK	0.921	0.907	0.723
CAPE/LK	0.885	0.875	0.614

Table 5 – Spearman's rank correlation coefficients between the conservation status (CR, EN, VU or LT) of each ecosystem type based on the quantification of habitat loss from the three land cover databases: the CAPE land cover map (CAPE), the NLC and the Little Karoo transformation map of (LK)

	BHUs	SANBI vegetation units	Little Karoo vegetation units
CAPE/NLC	1	0.817	0.374
NLC/LK	0.41	0.608	0.493
CAPE/LK	0.41	0.471	0.284

3.2 (ESRI, Redlands, CA, USA) on the original extent of ecosystem types, and the rank correlation between status categories of the three assessments of each ecosystem classification was measured (Spearman's rank correlation coefficients, R Development Core Team, 2007). The spatial overlap between threatened ecosystems (i.e. the three categories together) of the

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assessment based on Little Karoo vegetation map and Little Karoo land cover map (i.e. both finest maps of ecosystems and land cover, respectively) and the eight other assessments, was measured in pairs. This was performed by working on a grid of cells of 1 x 1 km. A grid of threatened cells was derived for each of the nine assessments. A cell was classified threatened when its centre was contained in an overlaying threatened ecosystem type (ArcView 3.2, ESRI, Redlands, CA, USA). The grid for the assessment Little Karoo vegetation map and Little Karoo land cover map were then overlaid by pairs with the eight other assessments. The number of common threatened grid cells (n_c) and the total number of threatened grid cells for each assessment (n_i) were counted. The formula $n_c/(n_1+n_2-n_c)$ was used as a measure of spatial overlap.

3. Results

3.1. Mapping habitat transformation

All three maps reveal some hotspots of habitat transformation across the Little Karoo region (Fig. 3). Habitat transformation by urbanisation, cultivation (including forestry plantations), severe overgrazing impacts and other severely-impacted unsustainable land use practices (e.g. mining) or high-density stands of invasive alien plant species is more highly concentrated along certain axes in the lower plains which are: in the vicinity of the perennial river systems of the Olifants, Gamka, Kammanassie and Little Leroux and along the national road N9 at the South border, in the Western half of the region, and around Montagu, in its Eastern half. The Little Karoo transformation map shows a more fragmented distribution of transformation than the NLC and the CAPE land cover, and further adds to the list transformation hotpots that correspond to the lower plains in the valley of the Groot and the Touws rivers. Areas classified as pristine in the Little Karoo transformation map were typically in rugged terrain. The three land cover databases showed high overlap for cultivated and urban areas (the similarity of agriculture and urbanisation patterns between the NLC and CAPE land cover for the CFR has also been stated by Rouget et al. (2003)). However, extensive tracts of lands mapped as severely degraded in the Little Karoo transformation map were overlooked in the CAPE land cover and largely underestimated in the NLC databases, where they were usually mapped as natural habitats. There were also situations, where some areas considered transformed in the NLC database (“degraded shrublands and low Fynbos”) were “pristine” areas in the Little Karoo transformation database. Assuming the Little Karoo transformation map is the most accurate database, there might be an overestimation of habitat

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Table 6 – Number of threatened ecosystems of each ecosystem classification based on the quantification of habitat loss from the three land cover databases: the CAPE land cover map (CAPE), the NLC and the Little Karoo transformation map of (LK) (The total number of ecosystem types per classification is: 26 BHUs; 40 SANBI vegetation units; and 369 Little Karoo vegetation units)

	BHUs (%)			SANBI vegetation units (%)			Little Karoo vegetation units (%)		
	CAPE	NLC	LK	CAPE	NLC	LK	CAPE	NLC	LK
CR	0 (0)	0 (0)	0 (0)	0 (0)	1 (2.5)	2 (5)	1 (0.3)	11 (3)	25 (6.8)
EN	0 (0)	0 (0)	0 (0)	1 (2.5)	0 (0)	0 (0)	2 (0.5)	11 (3)	29 (7.9)
VU	1 (3.8)	1 (3.8)	5 (19.2)	2 (5)	1 (2.5)	4 (10)	4 (1.1)	11 (3)	31 (8.4)
Total	1 (3.8)	1 (3.8)	5 (19.2)	3 (7.5)	2 (5)	6 (15)	7 (1.9)	33 (8.9)	85 (23)

transformation in certain areas of the NLC.

As a result of these differences, the quantification of habitat loss varied greatly across land cover databases. Overall, the Little Karoo region is currently estimated to have lost 6.1% of its original habitats with the CAPE land cover database and 10.6% by the NLC, while it is estimated at 24.7% by the Little Karoo transformation database. No ecosystem types had undergone complete transformation to the extent that they had no habitat remaining (Table 3), though, some of the Little Karoo vegetation units were estimated to have lost over 95% of their original extent with the NLC and the Little Karoo land cover.

Estimates of habitat loss within ecosystem types were statistically different between the three habitat loss assessments (Friedman tests, p-values << 1%; Table 3). The differences of estimates had a median value of about 10% of original extent between the Little Karoo and the CAPE land covers, 7% between the Little Karoo land cover and the NLC and 1% between the NLC and the CAPE land cover. In addition, these differences were the least consistent between the three assessments of the finest vegetation units of the Little Karoo vegetation map; Spearman's rank correlation coefficients between habitat loss estimates were overall high for BHUs and for SANBI vegetation types, but lower for the Little Karoo vegetation types (Table 4).

3.2. Mapping threatened ecosystems

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The number of threatened ecosystems ranged from about 2 to 23% of ecosystem types (Table 6), or, in terms of areal extent, from approximately 7 to 37% of the Little Karoo extant areas (Table 7). The patterns of threatened ecosystems (i.e. critically endangered (CR), endangered (EN) and vulnerable (VU)) were a direct manifestation of the concentration of habitat transformation more in certain localities than others. They, therefore, reflected the differences of habitat loss quantification between land cover databases (Fig. 5, 6, 7). The correlation between CR, EN and VU ecosystem types was generally low, in particular for the Little Karoo vegetation units (Table 5). The number of ecosystems categorised CR, EN and VU varied from assessments based on one land cover database to another, with the exception of threatened BHUs, that were invariably categorised VU (Fig. 5, 6, 7 and Table 6). In partcicular, 24.7% and 11.8% of Little Karoo vegetation units categorised least threatened with both the CAPE land cover and the NLC, were categorised EN and CR, respectively, from the Little Karoo land cover database.

Threatened Little Karoo vegetation units, identified with the Little Karoo land cover map, had a higher spatial overlap with threatened BHUS and SANBI vegetation units, respectively, when they were themselves, also identified with the Little Karoo land cover map. The spatial overlap was inferior to 9% when BHUS and SANBI vegetation units were identified with the CAPE land cover and the NLC; but, it was between 28 and 40% when they were identified with the Little Karoo land cover map. The total natural area classified as threatened was over 30% of lands with the Little Karoo transformation database, while it was 11 to 14% with the NLC and less than 10% with the CAPE land cover (Table 7). The number of ecosystem types classified as threatened was always higher where the threat assessment was performed with the Little Karoo land cover (Table 6). At least 15% of ecosystem types (up to 23% of Little Karoo vegetation units) were threatened with the Little Karoo land transformation map, but it was less than nine percent with both other land cover databases. In

Table 7 – Percentage of total extant area classified as threatened for each ecosystem classification based on the quantification of habitat loss from the three land cover databases: the CAPE land cover map (CAPE), the NLC and the Little Karoo transformation map (LK)

	CAPE	NLC	LK
BHUs	7.19%	11.54%	34.43%
SANBI vegetation units	7.29%	11.41%	37.48%
Little Karoo vegetation units	6.98%	14.38%	33.19%

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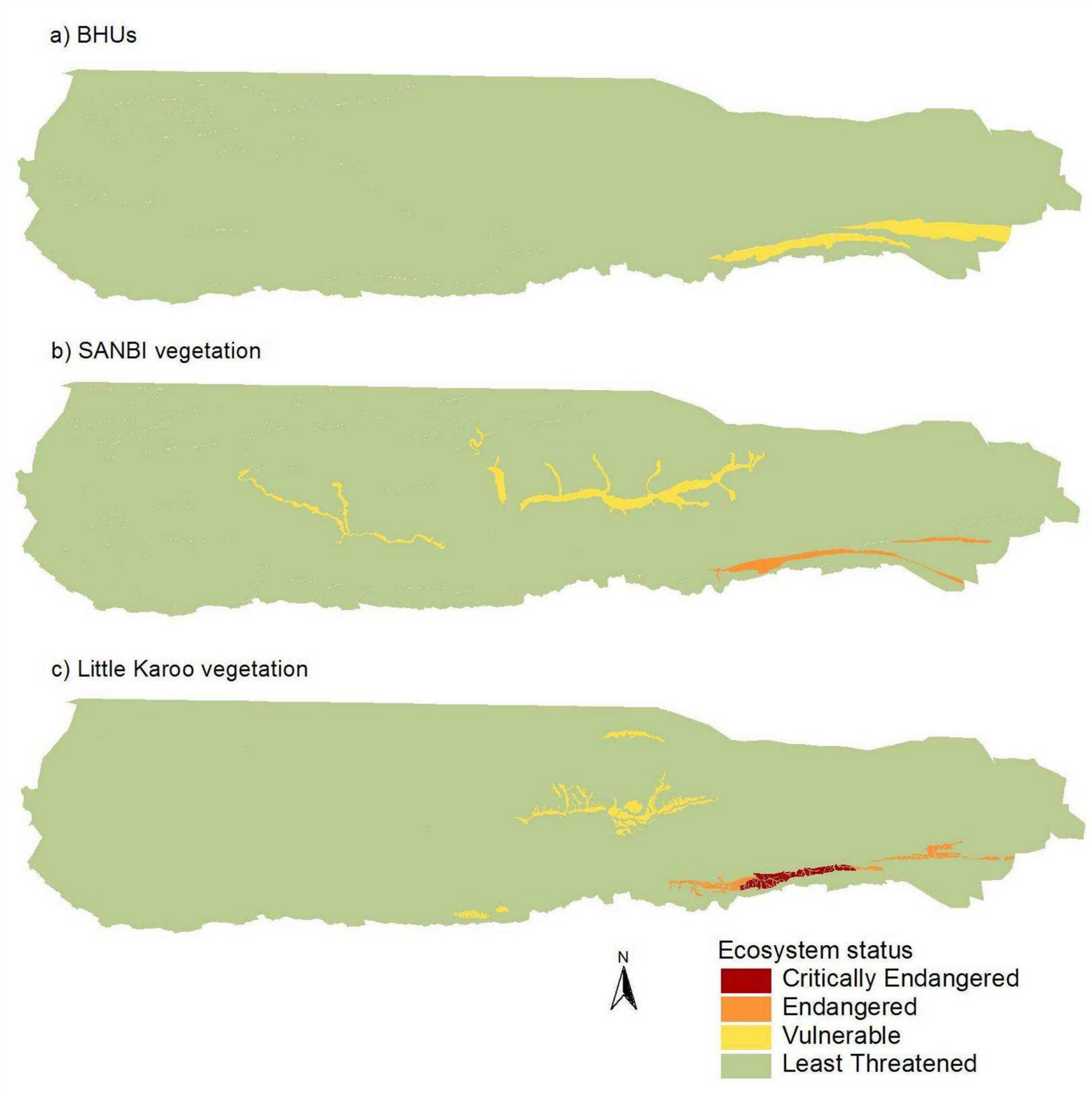

Fig. 5 – Patterns of threatened ecosystems, i.e. critically endangered, endangered and vulnerable, and least threatened ecosystems, displayed on original extents of ecosystem types for the three ecosystem classifications (Table 1), as obtained from the habitat loss estimation from the CAPE land transformation database (Cowling et al., 1999).

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particular, the threat assessment performed with the CAPE land cover database considerably underestimated threatened Little Karoo vegetation units (Table 6).

Ecosystem types assessed threatened based on one land cover database, were generally also included as threatened based on a database identifying a larger number of threatened ecosystems (i.e. threatened ecosystems based on the CAPE database were also generally threatened with the NLC and threatened ecosystems based on the CAPE or NLC databases were also generally threatened based on the Little Karoo database). In the case of Little Karoo

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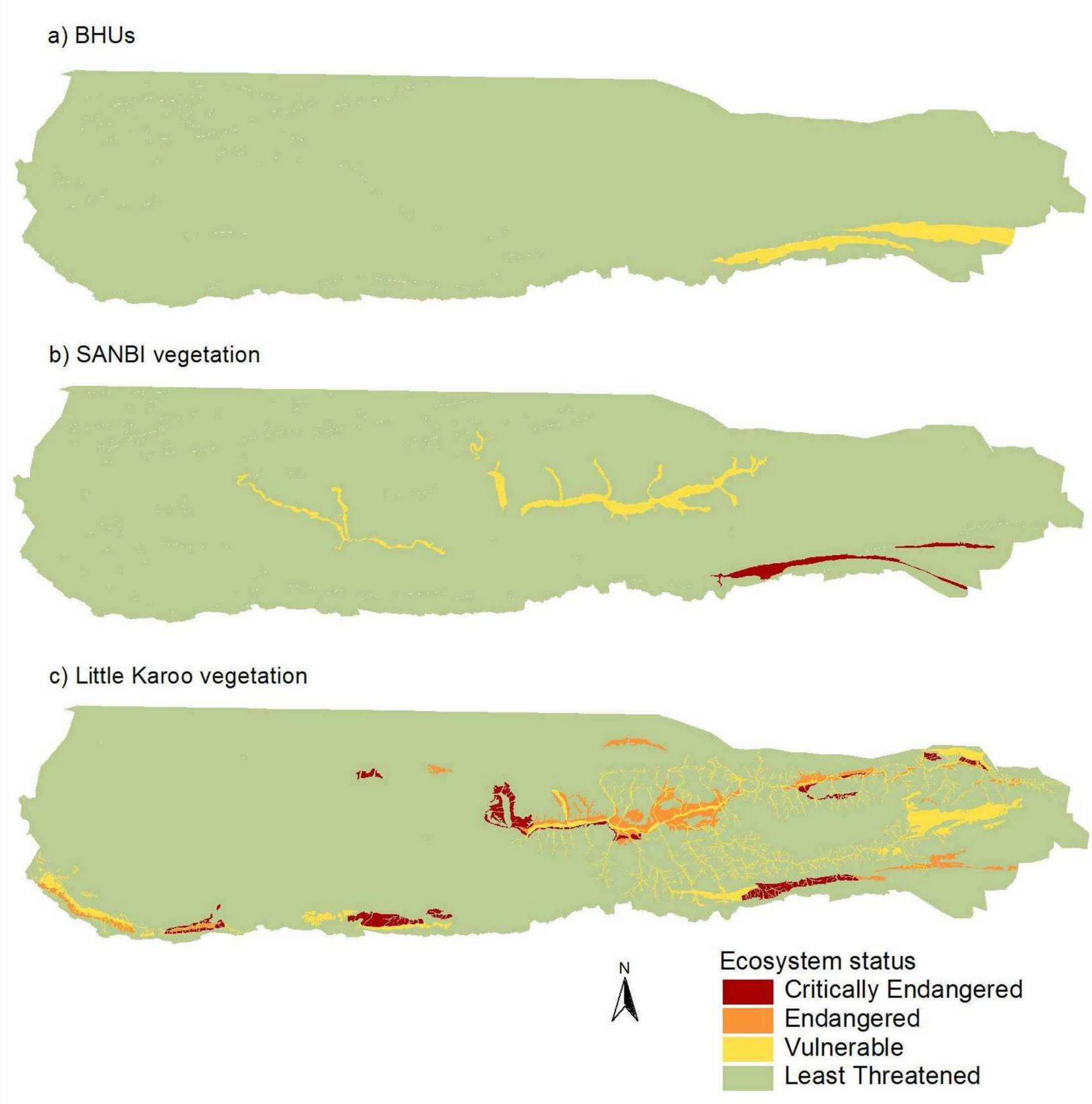

Fig. 6 – Patterns of threatened ecosystems, i.e. critically endangered, endangered and vulnerable, and least threatened ecosystems, displayed on original extents of ecosystem types for the three ecosystem classifications (Table 1), as obtained from the habitat loss estimation from the NLC (Fairbanks et al., 2000).

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vegetation units, there was a minority of cases where units were categorised at a lower endangerment level in the assessment based on the Little Karoo land cover database, compared to the NLC one.

4. Discussion

In a recent evaluation of the effective use of NSBA conservation products, Reyers et al.

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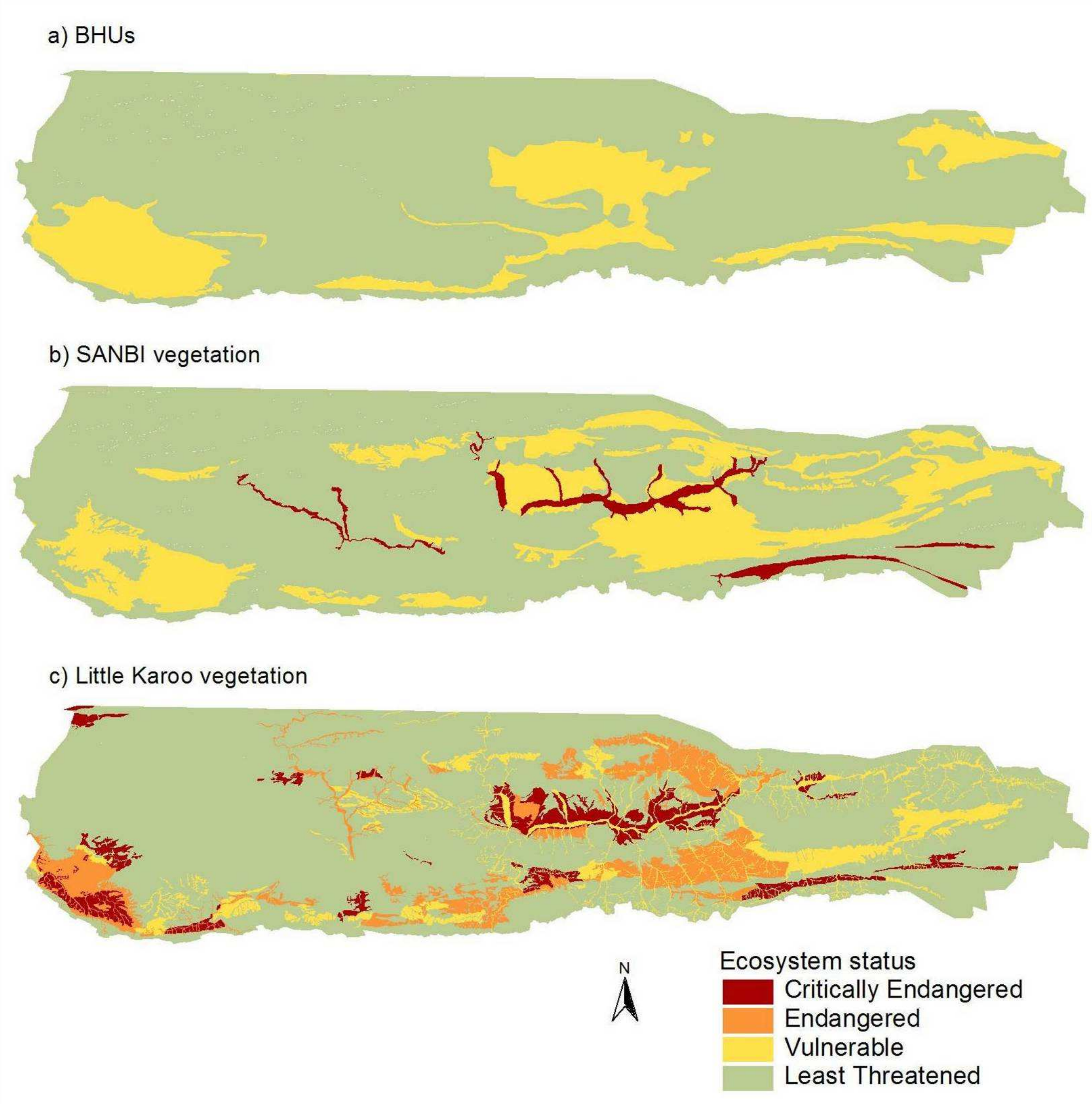

Fig. 7 – Patterns of threatened ecosystems, i.e. critically endangered, endangered and vulnerable, and least threatened ecosystems, displayed on original extents of ecosystem types for the three ecosystem classifications (Table 1), as obtained from the habitat loss estimation from the Little Karoo transformation map (Thompson et al., 2005).

(2007) identified that, in comparison to the maps of protection levels of ecosystems and priority areas, the conservation status map appears the most useful and popular product with implementing agencies, for it is easy to use and comprehend. This supports that the listing of threatened ecosystems under the South African Biodiversity Act (No. 10 of 2004; DEAT, 2004) has the potential to transmit a powerful conservation message and to serve as an effective implementation tool, in a similar manner to threatened species lists (Rodrigues et al., 2006). This potential of conservation status listing and mapping clearly needs to be fully

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exploited by the conservation community, given the gap that has to be bridged between the production of results from a conservation assessment and their implementation on-ground for conservation actions (Knight et al., 2006). The suitability of conservation implementation to exactly alleviate further biodiversity losses will partly rely on how accurately these maps depict the level of threat of each ecosystem type across a landscape. Yet, this study has shown the sensitivity of threat assessments, at ecosystem level, to the choice of land cover database (Fig. 5, 6, 7).

The identification of threatened ecosystems was more sensitive to the accuracy of habitat loss quantification than the resolution of ecosystem classifications. The endangerment of ecosystem types was significantly underestimated from the NLC and the CAPE land cover databases, quantitatively (i.e. fewer number of threatened ecosystems), but also qualitatively (i.e. different threatened category). Hence, it is important that risk assessments at ecosystem level are performed on accurate and detailed land cover mapping. Less than 11% of the Little Karoo region was estimated lost from the NLC and the CAPE land cover databases, while it was as much as 24.7% with the Little Karoo transformation map. The lack of congruence in patterns of endangerment was a direct implication of the underlying patterns of habitat loss. These differences in number and status of ecosystems between the coarse scale CAPE and NLC land cover databases and the fine scale Little Karoo data are mostly due to the mapping of areas transformed by overgrazing. This is similar to findings by Rouget et al. (2006) who found that using the Little Karoo land cover map provided a much lower estimate of Biodiversity Intactness than the NLC data because of the databases ability to map overgrazed and severely degraded areas.

Because resource limitations prevent the conservation of all areas of biological interest (Margules and Pressey, 2000; Sarkar et al. 2004), the foreseeable implication of habitat loss underestimation is that conservation attention and effort are driven away from ecosystems that are not identified as threatened or are not appropriate for ecosystems that were categorised in a lower threat status. But, the reverse would also be just as problematic, since an overestimation of habitat lost (as there might have been instances from the NLC) would also cause conservation-worthy remnants to be overlooked or that attention is concentrated on ecosystems unduly assigned a high threat level to the detriment of other more threatened ones in real fact.

The mapping of land cover from remotely sensed data is a delicate exercise that may lead to spatial and thematic misclassifications (Fairbanks and Thompson, 1996). This is particularly true of the mapping of land degradation (Rouget et al., 2006), and habitat degradation caused by overgrazing or chemical pollutants, and alien species stands were

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stated as being particularly problematic when generating the NLC and CAPE land cover databases (Lloyd et al., 1999; Fairbanks et al., 2000; Rouget et al., 2003). On the contrary, Thompson et al. (2005) designed a rigorous methodology to derive a more detailed mapping of habitat degradation by overgrazing in the Little Karoo region. This database contained extensive areas that were mapped as severely degraded where they were classified natural in both other ones.

It may be argued that these discrepancies are due to the artefact of using land cover databases of different ages, since a gap of almost ten years separated the data capture of the oldest (NLC) and the most recent (Little Karoo) ones (1996 to 2005). However, there are reasons to believe, that discrepancies were largely due to inaccurate mapping of severe habitat degradation in the older datasets, and not a result of recent land cover changes in the last ten years. Habitat degradation is primarily a result of the intense ostrich farming industry, the dominant activity throughout the region, established in the area in the 19th century (in Herling et al., in press; Hoffman and Ashwell, 2001). Most of the grazing impacts happened before the period studied here, and then relaxed due to less demand on ostrich farming in the area (C. Cupido, personal communication). Furthermore, Rouget et al. (2003) and Fairbanks et al. (2000) mention that the coding of overgrazed or sparsely vegetated areas was particularly challenging in the Karoo area for respectively the CAPE and the NLC databases. These factors suggest that the temporal parameter should be minor in the overall differences of land covers and can in majority be attributed to differences in mapping accuracy.

The use of fine-tuned ecosystem classifications is often constrained by limited time and finances (Rouget, 2003). Hence, a rather promising finding was that threatened BHUs and SANBI vegetation units showed a good spatial overlap with threatened Little Karoo vegetation units based on the Little Karoo land cover database (taken as the ultimate desired outcome), providing that the threat assessment was also performed with the Little Karoo land cover. Hence, this result suggests that where budgetary constraints impose the choice between developing a more detailed ecosystem map or a more detailed land cover map when a coarser version of each is available, priority should be given to mapping the latter.

The use of land cover data are crucial to effective conservation planning because they allow to identify or monitor some of the drivers of biodiversity loss (Firbank et al., 2003; Smith, 2003), habitat loss, degradation and fragmentation (Soulé, 1991; Wilcove et al., 1998). Rouget et al. (2006) further highlight the value of detailed land-cover data for conservation assessments. They state that the mapping of the Little Karoo land transformation data was not expensive, costing around $11 000, and rapid, taking less than a year to complete, and conclude that similar methods should be applied to other areas, thereby producing a very

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valuable contribution to conservation. These findings also strongly support this standpoint and similarly emphasise the need for reliable land cover databases for conservation assessments. In addition, a threat assessment performed only on habitat loss, like it was undertaken here, is insufficient because it does not take into account two major drivers of biodiversity loss, habitat fragmentation and degradation (Soulé, 1991). Reyers et al. (2007) suggests that this methodological flow is due to the lack of adequate land cover mapping. It is, for instance, likely that a reassessment of the Little Karoo region taking moderate habitat degradation into consideration will reveal more accurate patterns of endangerment given the extensive moderately degraded areas mapped in the Little Karoo transformation map. The availability of detailed land cover maps like the one produced by Thompson et al. (2005) would allow habitat degradation to be taken into account (Reyers et al., 2007).

5. Conclusion

The listing of threatened ecosystems on the model of the IUCN Red List of species can greatly assist conservation efforts. When addressed by a systematic approach, it has the advantage of being transparent and defensible. It has been demonstrated in this study, that the accuracy of systematic assessment of threatened ecosystems on the habitat loss criterion, is dependent to the quality of the data used. This is particularly relevant to land cover mapping. The increasing availability of remotely sensed satellite data and powerful computers should create increasing opportunities to derive accurate land cover information.

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Chapter 5

Conclusion

1. Key messages

This thesis presented three independent research components that, together, carry clear messages on the choice of data in systematic conservation assessments at different scales.

1. Judging from peer-reviewed articles, too much focus is still given to the use of species as biodiversity surrogates in conservation assessments (Chapter 2). In particular, assessments at large scales, were based almost exclusively on species data. It is a problem in the sense that these assessments were usually performed for a limited number of taxonomic groups (birds in particular) and often addressed one group at a time. In addition, the findings of Chapter 3 made apparent that species were not effective regional-scale biodiversity surrogates for the representation of other species, as well as ecosystems and processes at local scale. It, therefore, clearly is an inappropriate strategy to focus assessments entirely on limited data sets of species, if the objective is to maximise the representation of all biodiversity.
2. The low usage of biodiversity surrogates derived partly or solely on abiotic data (e.g. ecosystem types, topography) was relatively unexpected (Chapter 2), because there are reasons to believe that, for limitations in species datasets, real world conservation planning uses more of these surrogates, in particular at fine scale (Ferrier, 2002). Hence, it is possible that the review (Chapter 2) might have underestimated the use of community/assemblage and environmental surrogates, because the scientific literature does not provide a sample of works that is fully representative of real-world systematic conservation plans.
3. No single type of biodiversity surrogate, between ecosystems, species and processes, is sufficient to represent both others (Chapter 3). A combination of all three types of biodiversity surrogates is the basic requirement to the design of conservation networks for biodiversity conservation. Thus, a rather promising finding from the literature review (Chapter 2), was that fine-scale assessments were often based on combinations of different types of biodiversity surrogates and on socio-economic criteria. However, conserving species, ecosystems and processes can be land hungry (Chapter 3).

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4. Efforts to compile datasets on ecosystems, species and processes can be spared by selecting the surrogates in relation to the coarse filter/fine filter approach (Chapter 3). Furthermore, when conservation assessments are performed at regional and local scales, assessing the coarse filters at regional scale and the fine filters at the local scale should prove an adequate solution. The fine-scale mapping of the fine filters (i.e. imperilled species and ecosystems) could be restrained geographically, to areas impacted by habitat transformation, particularly the lowlands (Chapter 3 & 4). Hence, accurate land cover mapping is crucial to effective conservation assessments (Chapter 3 & 4).
5. Planning frameworks that adapt the choice of biodiversity surrogates to the spatial scale of the assessment, might be a very efficient way to pursue conservation assessments (Chapter 3). However, interestingly, this is currently not or little practised (Chapter 2). Protocols specifically taking the aspect scale/choice of surrogates into account should be more widely adopted and investigated.

2. Conserving how?

Conservation biologists generally agree on what they would ultimately like to see achieved: *biodiversity* conservation. However, approaches differ in two main ways. On one hand, some authors support the fact that conservation should target those features under most urgent need of extinction. This is a short-term conservation response that often focuses on threatened species, like it was the case of most species assessed at regional scale in Chapter 3. Hotspots of endemicity or richness are identified based on this premise. This approach could be described as being curative. Conversely, other authors adopt a more long-term perspective which implies that biodiversity is perceived more holistically. Networks of priority areas are identified to represent the full heterogeneity of patterns and processes (Chapter 3), to plan for representation and persistence (Cowling et al., 1999). This is performed in order to maintain a functional landscape, i.e. one that is persistent and resilient through time. Holistic approaches consider that land use changes will keep on occurring and that places have a high conservation value, even before they are transformed. This approach is both curative and preventive.

Approaches that are only curative are likely to be obsolete for many landscapes. Focusing efforts on safeguarding, restoring or recovering the most endangered features, was probably well suited to environments where habitat transformation was very localised, and biodiversity and landscape functionality decline was not escalating. The main drawback of such approaches is that they consider landscapes as static elements. However, everyday, land-

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use decision makers have to make decisions on developments: what can be allowed and where. Designing conservation networks for representation and persistence, will provide decision makers with the tools to make well-considered decisions in favour of biodiversity conservation through time. The fact that these conservation networks are extensive should be regarded as an unavoidable necessity (Chapter 3). Planning frameworks have to include carefully thought-out strategies and means to facilitate their effective implementation (Knight et al., 2006).

3. Recommendations for future study

3.1. Communicating better on research in conservation planning

Peer-reviewed articles are a prevailing communication means between researchers. They also contribute to educating postgraduate students. It is therefore important that the scientific literature gives a fair representation of real-world conservation planning. As stated earlier, there are reasons to suspect that ecosystem-type surrogates are more widely used than was revealed by the survey undertaken in Chapter 2. One possible explanation of this bias might be that conservation planning is performed mainly by a community of professionals, that is not research-focused, and that has neither the time nor the interest to publish its work. Another explanation might be that the criteria applied to the selection of submitted articles, are driven by interests that do not allow a representative sampling of real conservation planning efforts. While the former point might be impossible to remedy to, the latter clearly is a matter of journal politics.

Why should this be problematic? At least one reason was identified in this thesis. Ecosystems and processes are important biodiversity surrogates. However, few published articles communicate on these data derived on environmental variables. This lack of information hinders the accessibility to, and the exchange of experiences on, methods relative to their mapping and their application in systematic conservation assessments. At the same time, this may delay advances in a field of science that is under the intense pressure of urgency. The assessment of the effectiveness of conservation planning methods is difficult if published articles provide a sample that is not representative of what is actually being researched and/or applied.

A shortcoming of research in the use of biodiversity surrogates for conservation planning, is that experimentation is not, or hardly, replicable. This is due to limitations on the number of biodiversity features that can be tested in a same planning domain. Hence, it is

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often experiments undertaken by different teams of scientists from different regions and countries that, together, will allow rigorous conclusion to be drawn. Reyers and van Jaarsveld (2000) pointed out that a consensus on the effectiveness of biodiversity surrogates could not be reached, due to the lack of similarity in assessment techniques, and called for a standardisation of techniques. Hence, a foreseeable gain might be that research gets published even when experimentations are performed on similar methods, and lead, or not, to similar conclusions. Articles should not get published merely for the "novelty" of the techniques and approaches employed. It is a certain level of repetition that will create the replication required. This will enable the conservation planning community to mutually benefit from each other's accomplishments and mistakes more largely, and will allow more rapid progress in this field.

3.2. Spatially explicit conservation protocols

In brief, this thesis suggests two practical recommendations for the performance of systematic conservation assessments.

1. Perform assessments based on ecosystems + processes + species, selected in the manner characteristic to coarse and fine filters; if time and budget impose choices between biodiversity surrogates, always choose the coarse filters, and use the fine filters only locally for transformed areas;
2. Develop fine-scale and accurate land cover maps for habitat transformation, degradation and fragmentation; focus field work surveys and fine-scale mapping for imperilled species, communities and ecosystems (i.e. the fine filters) on these highly pressurised areas.

Kinght et al. (2006) provide similar recommendations. Some systematic conservation planning protocols already exist that integrate assessments at various spatial scales. The work performed by The Nature Conservancy (www.nature.org), the largest private land-protection organisation in the United States, is a good example. The Nature Conservancy also has a good methodology on how to select coarse and fine filters, and Noss (2003) provides a very helpful checklist for that as well. These works can be used to identify the coarse and fine filters to be assessed at regional and local scale.

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