Integrating ecosystem services into conservation planning in South Africa

by

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Declaration

By submitting this dissertation electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the owner of the copyright thereof (unless to the extent explicitly otherwise stated) and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

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ABSTRACT

Natural ecosystems provide many services that are crucial for sustainability and health of human society. Ecosystem services are the benefits people obtain from ecosystems (i.e. goods and services) and can be classified into provisioning (e.g. fibre, fuel wood); regulating (e.g. water and climate regulation); supporting (e.g. soil retention) and cultural (e.g. aesthetic value). The growing global human population and other threats place enormous stress on the natural environment reducing its capability to provide services. According to the Millennium Ecosystem Assessment, more than 60% of ecosystem services worldwide are being degraded or used unsustainably. The need to safeguard ecosystem services is therefore urgent.

Biodiversity underpins most ecosystem services, but the functional relationship between biodiversity and services is not well known. A wide range of strategies exist for safeguarding biodiversity, but no such approaches have been developed for ecosystem services. A key conservation strategy is the use of systematic conservation planning to identify priority areas where effort should be focused. There are calls for the inclusion of ecosystem services into conservation planning geared towards biodiversity. Ecosystem services have been used for many years as an additional rationale to justify biodiversity conservation and it is often assumed that conserving biodiversity will also conserve services. However, it is unclear how different facets of biodiversity relate to different services and to what extent conserving biodiversity will safeguard services.

This thesis addresses a range of issues relating to the integration of ecosystem services into conservation planning in South Africa. I first investigated the status of ecosystem services in conservation planning worldwide by reviewing the conservation planning literature from 1998 to 2005. Ecosystem services are clearly not adequately addressed in conservation assessments. A critical barrier preventing the inclusion of ecosystem services in conservation plans is the lack of spatially-explicit data. I developed a methodology for mapping ecosystem services in South Africa and mapped the distribution of five important ecosystem services (surface water supply, water flow regulation, carbon storage, soil retention and accumulation). Using the five services to examine relationships within services and between biodiversity revealed a lack of congruence between services and different levels of congruence with biodiversity features. However, including ecosystem services in a biodiversity assessment captured at least thirty percent of each of three services selected for the study. Nevertheless, a biodiversity plan may not necessarily capture adequate amounts of ecosystem services. Ecosystem services should be planned for explicitly instead of relying on biodiversity data. I identified priorities that met targets for five services in the grasslands of South Africa. This thesis provides new insights on planning for biodiversity and ecosystem services. The results have immediate applicability for conservation planning in South Africa.

Keywords: Conservation planning, conservation assessments, ecosystem functions, ecosystem processes, ecosystem services, natural capital, biodiversity, soil, water, carbon.

OPSOMMING

Natuurlike ekosisteme lewer baie dienste wat van uiterste belang vir die volhoudbaarheid en gesondheid van die mensdom is. Ekosisteemdienste behels die voordele wat mense uit ekosisteme kry, en kan as verskaffende (bv. vesel, brandhout), reguleerende (bv. water en klimaat regulasie), ondersteunende (bv. grondbewaring) en kulturele dienste geklasifiseer word. Die toenemende globale menslike bevolking en ander bedreigings is besig om ontsettende spanning op die natuurlike omgewing te plaas, wat die vermoë om dienste te lewer verminder. Volgens die Millennium Ekosisteem Waardering is meer as 60% van ekosisteemdienste wêreldwyd of gedegradeer of besig om op ‘n onvolhoudbare wyse gebruik te word. Die noodsaaklikheid om ekosisteemdienste te beskerm is dus dringend.

Die meerderheid van ekosisteemdienste word ondersteun deur biodiversiteit, maar die verwantskappe tussen biodiversiteit en dienste is nie goed bekend nie. Daar bestaan ‘n wye reeks strategieë om biodiversiteit te bewaar, maar daar is nog geen vergelykbare benaderings vir ekosisteemdienste nie. Die gebruik van stelselmatige bewaringsbeplanning om prioriteite te identifiseer, is ‘n sleutel bewaringstrategie. Daar is beroepe om ekosisteemdienste by bewaringsplanne om biodiversiteit te bewaar, in te sluit. Ekosisteemdienste is al vir baie jare as ‘n bykomende rede voorgstel om biodiversiteit te bewaar en dit word dikwels aanvaar dat die bewaring van biodiversiteit ook die bewaring van ekosisteemdienste insluit. Dit is egter steeds onduidelik hoe verskillende aspekte van biodiversiteit koppel met ekosisteemdienste, en tot watter mate die bewaring van biodiversiteit ook dienste sal beskerm.

Hierdie tesis spreek probleme aan wat verband hou met die wyse waarop ekosisteemdienste by bewaringsbeplanning in Suid-Afrika ingesluit kan word. Ek het eerstens die status van ekosisteemdienste in bewaringsbeplanning wêreldwyd ondersoek deur die literatuur oor bewaringsbeplanning tussen 1998 en 2005 te hersien. Ekosisteemdienste word duidelik nie voldoende aangespreek in bewaringskattings nie. Die insluiting van ekosisteemdienste in bewaringsplanne word verhinder deur die gebrek aan ruimtelik-beduidende data. Ek het ‘n metode ontwerp om ekosisteemdienste in Suid Afrika te karteer, en het die verspreiding van vyf belangrike ekosisteemdienste (oppervlak water lewering, water vloei regulasie, koolstof berging, grondbewaring en opberging) gekarteer. ‘n Vergelyking van die verhouding tussen

die vyf dienste en biodiversitiet het daarop gewys dat daar ‘n gebrek aan ooreenstemmigheid tussen dienste en verskillende vlakke van ooreeenstemmigheid met biodiversiteitskenmerke was. Deur ekosisteemdienste in biodiversiteitskattings in te sluit, was dit tog moontlik om ten minste dertig persent van elk van drie geselekteerde dienste te vang. ‘n Biodiversiteit plan sal nietemin nie noodwendig genoeg van die ekosisteemdienste vang nie. Ekosysteen dienste behoort uitdruklik voor beplan te word, in plaas daarvan om net op biodiversiteit data te vertrou. Ek het prioriteite, wat teikens vir vyf dienste in die grasvelde van Suid Afrika ontmoet het, geidentifiseer. Hierdie tesis verskaf nuwe insigte oor beplanning vir biodiversiteit en ekosisteemdienste. Die resultate sal onmiddellike gebruik in bewaaringsbeplanning in Suid Afrika hê.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1.
GENERAL INTRODUCTION

1.1 Background

1.1.1 What are ecosystem services?

Natural ecosystems provide many services that are crucial for the sustainability and health of human society (Alcamo et al., 2003; Chiesura and de Groot, 2003). Several definitions and classification exist for ecosystem services (Daily, 1997; de Groot et al., 2002; Swift et al., 2004). The Millennium Ecosystem Assessment defines ecosystem services as the benefits that people obtain from the ecosystems, and classifies them into provisioning (e.g. fibre, fuel wood); regulating (e.g. water and climate regulation); cultural (e.g. recreational and cultural benefits); and supporting (e.g. nutrient cycling, soil retention) services (MA, 2003). However, Wallace (2007) suggests that regulatory and supporting services as classified by MA are means to achieving provisioning and cultural services. He proposed a new classification of ecosystem services whereby the services are described in terms of the composition and structure of particular elements (expressed as assets) and classified according to the specific human values they support. This proposed classification is currently under debate (Costanza, 2008; Fisher and Turner, 2008; Wallace, 2008).

1.1.2 Human dependence on ecosystems services

All over the world natural ecosystems perform fundamental life-support services without which human civilization would cease to thrive (Daily, 1997). Ecosystem services support more than one billion people living in extreme poverty worldwide (World Bank, 2006). Many communities around the world depend on forests for a wide array of timber and non-timber products such as fuel-wood, construction timber, carving wood, medicinal plants and animals, edible herbs, fruits, thatch grass and honey (Shackleton et al., 2004). In South Africa, some 57 different plants have been recorded for use in herbal remedies during pregnancy and childbirth (Veale et al., 1992). About 45 plant species from 24 families surveyed are used for veterinary purposes (Van der Merwe et al., 2001) and about 500 species of medicinal plants are known as commercialized trade products in South Africa (Hoareau and DaSilva, 1999).

As such, the livelihoods of most people in rural areas of South Africa, especially those in the former "homeland areas", depend on natural woodland and forest (Shackleton et al., 2001). Apart from the many harvestable products supplied by the forest, woodlands also store large amounts of carbon (Winjum et al., 1993).

1.1.3 Threats facing ecosystems services

The growing global human population and the subsequent increase in urbanization, agriculture, mining and other threats place enormous stress on the natural environment. In 2001, the World Resources Institute reported a widespread decline in the condition of the world's ecosystems (WRI, 2001). Following this report and others, the Millennium Ecosystem Assessment, a four-year international effort to assess the capacity of ecosystems to provide the services needed to support human well being and life on earth was commissioned (http://www.millenniumassessment.org). According to the MA (2005), over 60% of the services assessed across the globe are being degraded or used unsustainably with the potential to become more degraded in the first half of this century. This was seen as an important potential barrier to the achievement of the Millennium Development Goals (http://www.un.org/millenniumgoals, see box 1) (MA, 2005). These findings have brought to light the need to identify and manage areas that are important for the provision of these services (Balvanera et al., 2001; Kremen and Ostfeld, 2005; Chan et al., 2006).

Box 1.

UN Millennium development goals

1. *Eradicate extreme poverty and hunger*
2. *Achieve Universal primary education*
3. *Promote gender equality and empower women*
4. *Reduce child mortality*
5. *Improve maternal health*
6. *Combat HIV/AIDS, malaria and other diseases*
7. *Ensure environmental sustainability*
8. *Develop a global partnership for development*

The need to safeguard ecosystem services is urgent. Areas that are important in providing services have to be carefully managed. Successful management of areas important for the delivery of ecosystem services requires detailed information on many facets of ecosystem services including: the ecosystem components responsible for their provision (ecosystem service providers), their distribution, threats facing them and how these are inter-related, and an understanding of how the threat factors operate at different scales of space and time (Kremen, 2005; Kremen and Ostfeld, 2005). Such information, especially that relating to scales relevant for local management, are widely scattered, making it difficult to define objective strategies for appropriate management of these services (Balvanera et al., 2001, see chapter 2 for review). This has lead to calls to: increase our understanding of the ecology of ecosystem services (Kremen, 2005) and develop proper planning approaches for ecosystem services (Balvanera et al., 2001; Singh, 2002). An important strategy relating to safeguarding ecosystem services involves their inclusion into conservation planning.

1.1.4 What is conservation planning?

Conservation planning is a branch of conservation biology that seeks to identify spatially explicit options for the preservation of biodiversity (Margules and Pressey, 2000). In many countries, the selection of areas for conservation was historically driven more by socio-economic and political issues (Pressey, 1994). This often resulted in an unrepresentative protected area system (Rebelo, 1997; Reyers et al., 2001; Rouget et al., 2003a) biased towards unproductive and economically marginal landscapes (Turpie et al., 2003). Scoring systems were developed in the 1970s (Margules and Usher, 1981; Terborgh and Winter, 1983; Purdie et al., 1986; Smith and Theberge, 1986; Margules et al., 1988) to provide an explicit and rational basis for selecting conservation areas. Scoring systems rated natural areas against several criteria (see Dunn et al., 1999) to provide an overall indication of their conservation value within homogeneous biogeographic units that serve as a planning domain. However, they did not identify conservation areas in an efficient and effective manner (Pressey, 1997).

The minimum-set approach introduced in the 1980s was aimed at identifying the smallest area that conserved the greatest number of species. The requirement for a species-rich set, and not just individually-rich areas, suggested an alternative assessment based on a complementary-areas approach, which identified whole systems of complementary areas which collectively achieved some conservation goal (Kirkpatrick, 1983; Margules et al., 1988). Also in the

1980s, the utilisation of expert knowledge was common, as many groups, especially international conservation organizations, advocated and used expert workshops where conservation areas were identified using the experience of informed participants from a wide range of taxonomic, ecological, evolutionary and socio-economic disciplines (Prance, 1983; Olivieri, et al., 1995; Rodriguez and Young, 2000). The expert approach has been criticized because of the biases associated with experts' uneven knowledge of regions and taxa (Maddock and Samways, 2000). However, the expert approach has been recommended for use alongside other complementarity based approaches (Cowling et al., 2003).

The aim of conservation areas is mostly to separate them from threats facing biodiversity or bring down the threat level and to ensure the persistence of biodiversity. To achieve this, systematic conservation planning was developed (for a review see Margules and Pressey, 2000). Systematic conservation planning identifies conservation areas, which must be representative of the full variety of biodiversity, ideally at all levels of organization and ensure its persistence (Margules and Pressey, 2000). Systematic conservation planning has several distinctive characteristics: it requires biodiversity features that can be mapped spatially and used in the planning process, based on goals translated as targets, recognition of the extent to which goals have been met and uses explicit methods for locating and designing new reserves (Margules and Pressey, 2000).

1.1.5 Conservation planning and ecosystem services

Much effort in systematic conservation planning has gone into developing decision-support systems, which are efficient at capturing the biological importance of different areas (Moore et al., 2004). Broad goals for the identification of conservation areas are to maintain biodiversity pattern and ensure its persistence (Balmford et al., 1998; Margules and Pressey, 2000) through the inclusion of ecological and evolutionary processes (Desmet et al., 2002; Rouget et al., 2003a). Many conservation initiatives, particularly in the tropics, focus on narrow interpretations of biodiversity, ignoring ecosystem services, though conserving biodiversity is justified on the ground that it contributes to ecosystem services (Singh, 2002). In a study carried out in South Africa, less than 5% of conservation assessments between 1999 and 2003 have considered ecosystem services. However all of the studies do consider the intrinsic value of biodiversity (i.e. species and habitats) without directly taking into account ecosystem services rendered by maintaining natural habitat e.g. production of clean water (Rouget and Egoh, 2003).

Most of the data used for conservation planning are species distribution records and land types (Brook et al., 2004). Targets for species take into consideration minimum viable population size, elements of diversity, rareness, endemism, vulnerability, and species of special concern (Cowling et al., 2003; Von Hase et al., 2003). Targets for habitats are sometimes based on the species area-relationship (Desmet and Cowling, 2004). Throughout the planning process parameters like contribution to water purification, carbon sequestration, pollination, medicinal value or timber and fuel wood potential are often ignored. Recent studies suggest that further improvements in systematic conservation planning are more likely to come from measuring and integrating socio-economic data and other considerations than from focusing exclusively on refinement of biological criteria (Moore et al., 2004; Knight and Cowling, 2007). Identification of areas for conservation should include both biodiversity and human related functions.

1.1.6 The link between ecosystem services and ecosystem functions

Ecosystem services are also referred to as ecological services, ecological functions, environmental services and environmental functions. Daily (1997) defines ecosystem services as “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life”. De Groot (1992) defines ecosystem function as “the capacity of natural processes and components to provide goods and services that satisfy human needs directly or indirectly”. Costanza et al. (1997) refer to ecosystem functions as the habitat, biological or system properties or processes of ecosystems. Ecosystem functions therefore include biodiversity pattern and processes plus other abiotic (see Figure 1.1) components while ecosystem services are the benefits that people derive from these functions.

Ekins (2003) separates environmental functions into “functions of” and “functions for”. The “functions of” are those that maintain the integrity of natural systems in general and ecosystems in particular. The “functions for” are those that provide direct benefits to humans. Some ecosystem functions are therefore the mechanisms through which ecosystems provide goods and services that benefit humans. The use of ecological processes is common in conservation planning and refers to the functions that ensure the persistence of biodiversity

(Rouget et al., 2003a), for example riverine corridors (important for animal movements) and upland-lowland gradients (considered as important for plant and animal lineages). Therefore ecosystem processes as used in this thesis will refer to functions of ecosystems related to biodiversity persistence.

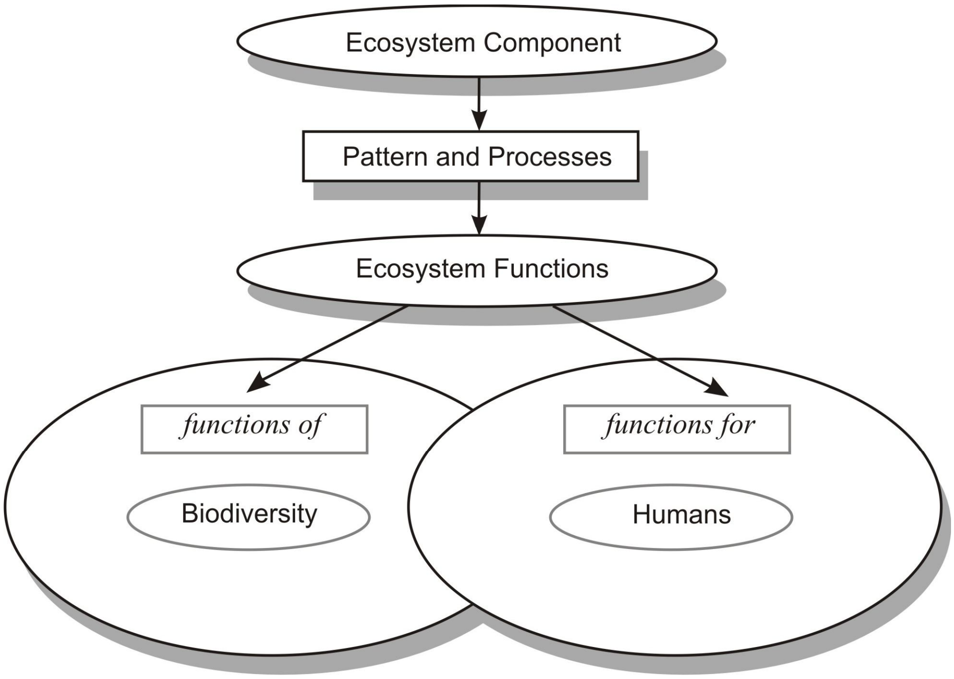

Figure 1.1: An illustration of ecosystem functions and how they relate to biodiversity and ecosystem services.

1.1.7 Relationship between biodiversity and ecosystem services

There is increasing debate on the links between biodiversity and ecosystem functions and services. Some scientists support a positive relationship between biodiversity and the persistence of ecosystem functions (see Swift et al., 2004). This is true to the extent that biodiversity as a supporting function is the basis for most ecosystem functions that provide services. The refugium function (de Groot et al., 2002) which deals with the maintenance of biodiversity and ecological process, is just one of the many ecosystem functions listed as important to humans. However, there is evidence to show that areas important for biodiversity are not necessarily important for delivery of services. In fact, many freshwater ecosystem services decline with an increase in species diversity (Singh, 2002).

Many studies have reviewed the link between biodiversity and ecosystem functions (see Lacroix and Abbadie, 1998; Schwartz et al., 2000; Diaz and Cabido, 2001; Swift et al., 2004; Srivastava and Vellend, 2005). While some studies claim there is little or no relationship, others have shown a positive relationship. Many of the studies that have shown a positive relationship between biodiversity and ecosystem functioning were not carried out in natural environments. Schwartz et al. (2000) studied 20 surveys that reported a positive relationship between species diversity and ecosystem functioning and 10 out of 13 studies evaluated observed a relationship at very low levels of diversity (7 of the 20 studies could not be evaluated). The nature and strength of the link between diversity and ecosystem functioning varies across spatial scales, from local plots to regional gradients (Diaz and Cabido, 2001). Lacroix and Abbadie (1998) conclude that the relationship between species composition, stability and ecosystem function needs to be investigated at a variety of spatial and temporal scales.

1.1.8 Will conservation plans geared toward biodiversity capture ecosystem services?

While the debate on the relationship between biodiversity and ecosystem functions/services is ongoing, the most important question for safeguarding ecosystem services will be to what extent biodiversity features or priority areas co-occur with ecosystem services. Until now, it is not clear whether (or to what extent) biodiversity priority areas capture individual ecosystem services or their priority areas. Chan et al. (2006), Turner et al., (2007) and Naidoo et al. (2008) investigated the special concordance between biodiversity priority areas and ecosystem services and found varied results at different scales. The results suggest concordance in some instances between areas important for biodiversity and those important for ecosystem services. Such findings highlight the need to continue the investigation in order to understand what ecosystem services can be captured by a conservation plan geared towards biodiversity and which ones need conservation strategies separate from those employed for biodiversity.

In a forum for conservation planners held in January 2005 at the uKhahlamba Drakensberg Park in KZN, South Africa, the integration of ecosystem services into conservation assessments was cited as a major opportunity and challenge in many of the presentations (http://www.sanbi.org/biodiversity/planning.htm). Although participants agreed that this could boost implementation of conservation plans because implementation agencies could use ecosystem services to make a case for biodiversity, this potential is constrained by a general lack of data on services and knowledge about how best to use services in these plans and their implementation. Further constraints highlighted include the increasing resource requirements of adding ecosystem services to already resource demanding conservation plans (Frazee et al. 2003), as well as the mixing of intrinsic and utilitarian values in conservation planning (McCauley, 2006). At the same time the opportunities offered by the inclusion of ecosystem services in conservation planning and the potential provided by existing approaches and tools from conservation planning make it important to explore these opportunities, making explicit the strengths and weaknesses of this integrated planning. The studies reported on in this thesis set out to explore the implications of including ecosystem services into conservation plans. This is done through the investigation of techniques to map and quantify services, approaches to include them into conservation planning, as well as an evaluation of the costs and other constraints of doing so.

South Africa is an appropriate place to carry out these studies because of its heterogeneous landscape and richness in biodiversity. The country has been the focus of much conservation planning and biodiversity research (van Jaarsveld et al., 1998; Reyers et al., 2001; Cowling et al., 2003), as well as agricultural and hydrological research (Schulze, 1997; Schoeman et al., 2002). Many good biophysical databases and assessment techniques are available (Balmford, 2003). In addition, the Southern African Millennium Ecosystem Assessment included South Africa in its assessment of ecosystem services (Scholes and Biggs, 2004; van Jaarsveld et al., 2005), initiating the collation of new databases on ecosystem services. A large proportion of South Africans live in rural areas on private commercial farms (1.5 million households) and the communal lands of the former homelands (2.3 million households) (Van Horen and Eberhard, 1995). Most of the rural people survive as subsistence farmers and depend on the natural area for collecting of resources.

1.2 Study Area

The studies in this thesis were carried out in South Africa at the national, regional and local scales (see Figure 1.2). South Africa, covering an area of about 1.22 million km^2, is extraordinarily rich in biodiversity, and is home to seven biomes and about 22000 plant species, about 10% of the world's flora (Coetzee et al., 1999) in 1% of the earth's total land surface (Driver et al., 2005). The country contains three biodiversity hotspots: the Succulent Karoo, Cape Floristic Region and Maputoland-Pondoland-Albany hotspots (Mittermeier et al., 2005), areas of high endemism and threat.

Threats facing biodiversity and ecosystem services include urban expansion, agriculture, invasive alien plants, overgrazing and mining. According to Fairbanks et al. (2000), approximately 30% of the fynbos biome is transformed and a further 10% of the savanna and 26% of the grassland biomes are currently degraded or transformed by human land uses. In the Cape Floristic Region alone (about 87,892 km^2 in extent), 25.9% of the surface area is transformed by agriculture and 1.6% by alien plants (Rouget et al., 2003b). These land-use practices and land cover changes, together with climate change, threaten biodiversity and ecosystem services in the country (Bohensky et al. 2004; Rouget et al. 2006; Reyers et al. 2007). For example, invasions by alien plant species displace indigenous species populations, transform the structure and function of ecosystems and change the levels of ecosystem services significantly. These plants consume more water than the country's indigenous vegetation, especially where dense stands of tall woody plants (trees and shrubs) replace natural grassland or shrubland (van Wilgen et al., 1996).

In response to these threats, conservation initiatives in South Africa are developing rapidly. Examples include the Working for Water poverty relief alien eradication program (van Wilgen et al., 1996), the Biodiversity and Wine Initiative which seeks to mainstream biodiversity considerations within the wine industry (www.bwi.co.za), and a large number of conservation plans spearheading the investment of conservation resources in spatially explicit priority areas (see Pierce et al., 2002) . Over 40 conservation planning initiatives were carried out between 1990 and 2003 in South Africa. About 10 of these were major projects (or bioregional programs) with many sub studies (e.g. the Cape Action Plan for People and the Environment; Cowling et al., 2003). This wealth of conservation planning, along with good funding, a strong research sector, capable implementing institutions, and major development needs, has seen significant advances in the country in terms of data sets, techniques, capacity and research in the field (Reyers et al., 2008).

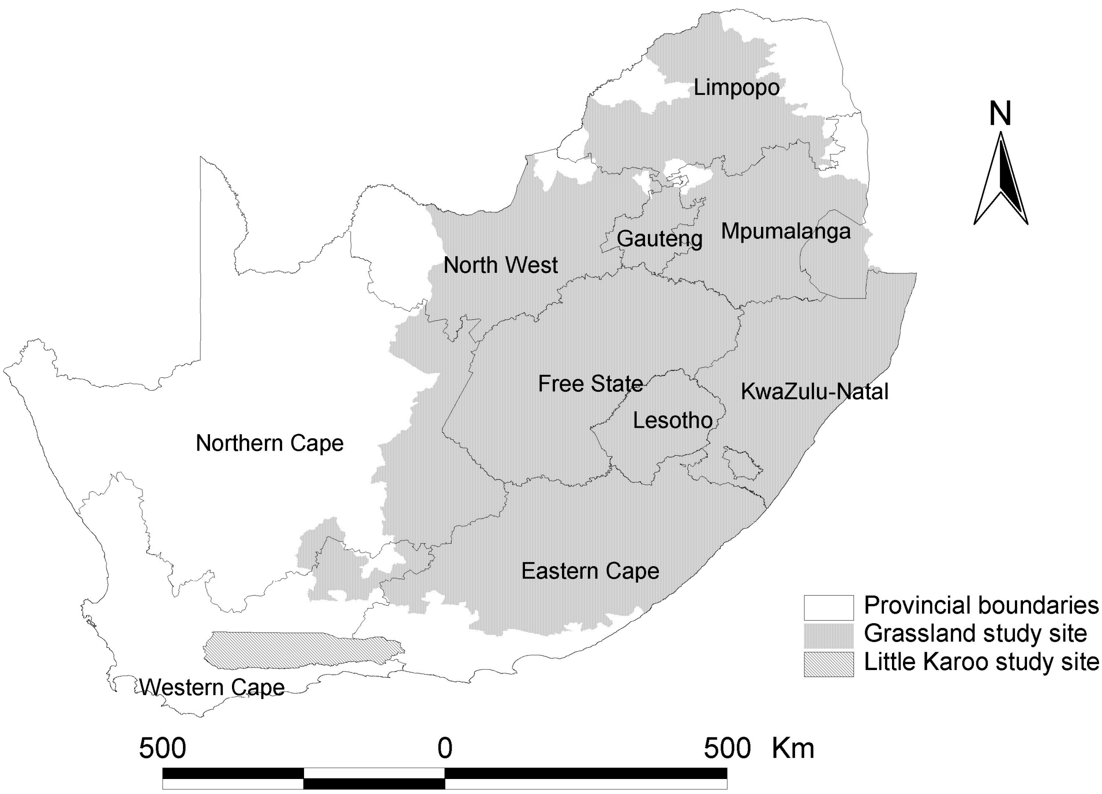

Figure 1.2: Map of South Africa, showing the nine provinces and the location of two regions for which detailed studies were undertaken: the grassland biome (Chapter 5), and the Little Karoo (Chapter 6). Chapter 3 and 4 conducted at the national scale

1.3 Key research questions and chapter breakdown

Cowling et al. (2008) proposed a framework for mainstreaming ecosystem services for implementation (see Appendix A) which consists of three phases: the assessment, planning and implementation phases. The studies in this thesis fall within the assessment phase and Figure 1.3 represents the position of each component of this thesis in the Cowling et al. (2008) framework, providing examples of questions that need answering at each stage and how these fit within the whole framework.

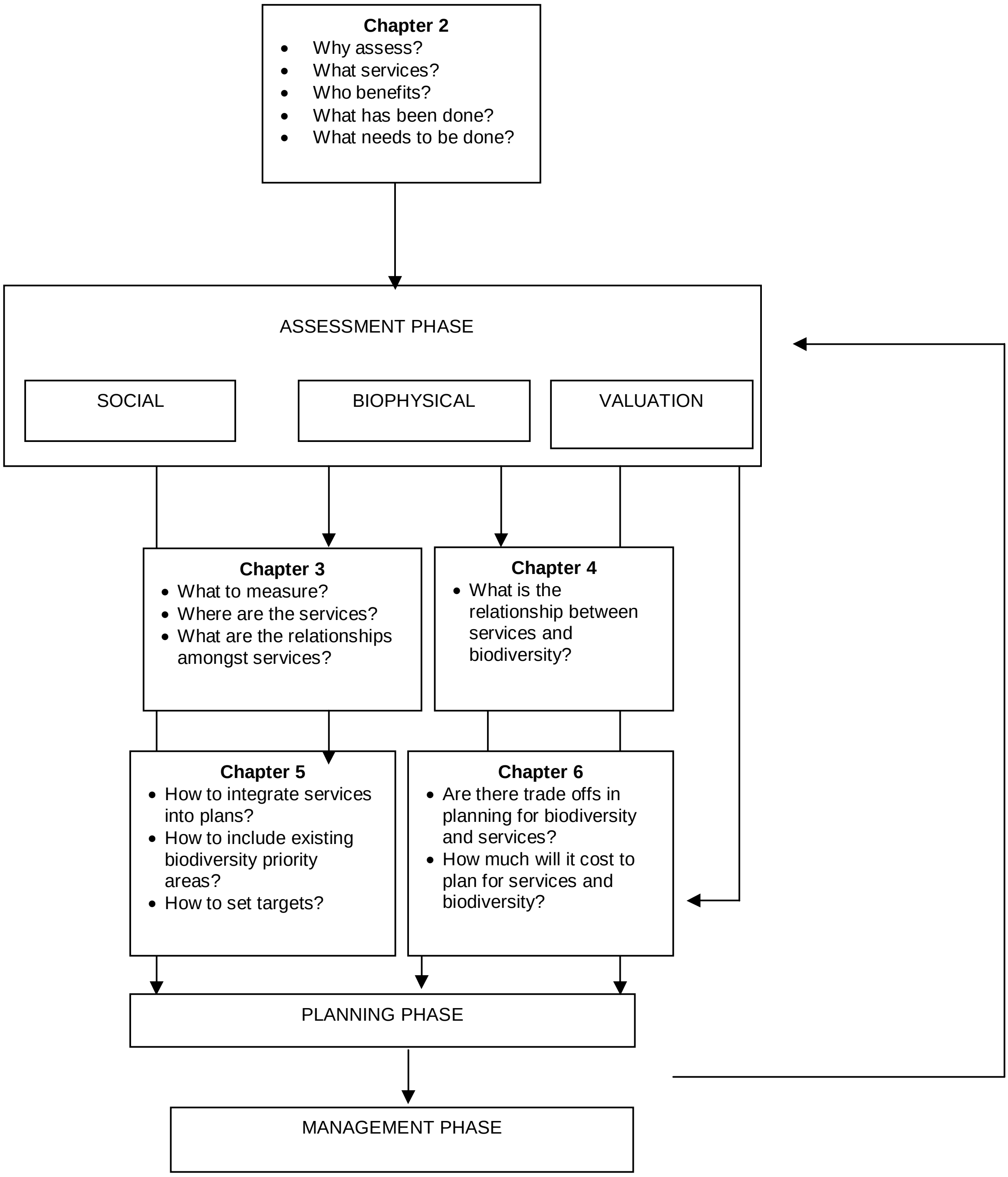

Figure 1.3: A schematic overview of the chapters in this thesis within a framework for mainstreaming ecosystem services for implementation as described in Cowling et al. (2008).

1.3.1 What is the status-quo of ecosystem services in conservation planning globally?

Chapter 2: Integrating ecosystem services into conservation planning: a review

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BE designed the study with support from BR, MR and RMC. BE collected data and did the analysis. BE, wrote the paper with support from BR and MR. ASJ provided input to the manuscript.

The aim of this chapter was to understand the status-quo of ecosystem services in conservation planning. It answers questions relating to whether ecosystem services are included in planning exercises, what services are included, and how. The key challenge in this chapter was to separate studies that have included services from those that did not. Many studies use ecosystem services as a rationale for biodiversity conservation, rather than as component of the conservation plan. The terms ecosystem services, processes and functions were also used interchangeably and it was necessary to have a proper definition and clear understanding of ecosystem services in order to tease the studies apart. The chapter ends with a discussion on the opportunities and constraints for including ecosystem services in conservation assessments and set the scene not only for the rest of this thesis, but also for the development of the operational framework for mainstreaming ecosystem services into management outlined in Cowling et al. (2008).

1.3.2 How do we map ecosystem services and what is their distribution in South Africa?

Chapter 3: Mapping ecosystem services for planning and management

This chapter is published as: Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., van Jaarsveld, A.S., 2008. Mapping ecosystem services for planning and management. Agriculture, Ecosystems and Environment 127, 135-140.

BE designed the research in consultation with BR and MR. BE collected data and did the analysis. BE wrote the paper. MR, BR, DCM and DMR provided input into the final manuscript.

A critical barrier to planning for or including ecosystem services into conservation assessments is the lack of spatially-explicit data on ecosystem services. The main aim of this chapter was to develop a methodology for mapping ecosystem services and to apply this in South Africa. Also of importance was the need to evaluate the spatial distribution of ecosystem services and to investigate the suitability of each mapped service as a potential surrogate for all other services that were considered. The main challenge in this chapter was the selection of appropriate proxies for mapping each service. The chapter ends with a discussion on the implication of the findings for managing ecosystem services in South Africa.

1.3.3 Do ecosystem services co-occur with biodiversity in South Africa?

Chapter 4: Spatial congruence between biodiversity and ecosystem services in South Africa
This chapter is submitted and accepted as: Egoh, B., Reyers, B., Rouget, M., Bode, M., Richardson, D.M. Spatial congruence between biodiversity and ecosystem services in South Africa. Biological Conservation.

BE designed the study in consultation with MR and BR. BE did the analysis with assistance from MB in statistical analysis. BE wrote the paper. MR, BR, MB and DMR provided input into the final manuscript.

An important question related to the safeguarding of ecosystem services is whether they co-occur in space with elements of biodiversity, and whether conserving these elements of biodiversity would safeguard the ecosystem services. In this chapter, the spatial congruence between biodiversity and ecosystem services was examined using three different, but commonly used methods (correlations, overlap and co-incidence analysis). The services mapped in chapter 2 were used to investigate the spatial relationship between ecosystem service hotspots and biodiversity hotspots. The study in this chapter concludes by discussing the biodiversity features that aligned with services.

1.3.4 How can we identify priority areas for ecosystem services?

Chapter 5: Identifying priority areas for ecosystem services in the grassland biome of South Africa. This chapter is to be submitted as: Egoh, B., Rouget, M., Reyers, B., Richardson, D.M., In prep. Identifying priority areas for ecosystem services in the grassland biome in South Africa. African Zoology.

BE designed the study in consultation with MR, BR and DMR. BE did the analysis. BE wrote the paper. All authors provided input into the final manuscript.

This chapter makes the first attempt to identify priorities for ecosystem services in the grassland biome of South Africa and to align the goals of biodiversity conservation with those of ecosystem services. Here, maps of ecosystem services (some extracted from chapter 2, some newly developed) were used to generate priorities for ecosystem services. Several planning scenarios were developed to examine the influence of ecosystem service targets and also to align the outputs with already identified terrestrial and freshwater biodiversity priority areas. The chapter ends with a discussion of the benefits of including biodiversity priorities into ecosystem service planning.

1.3.5 How can we plan for both biodiversity and ecosystem services and what are the cost implications?

Chapter 6: Safeguarding biodiversity and ecosystem services in the Little Karoo: synergies and trade-offs

This chapter is to be submitted as: Egoh B., Reyers, B., Carwardine, J., Bode, M., O'Farrell, P., Wilson, K., Rouget, M., Possingham, H., Cowling, R.M. Safeguarding biodiversity and ecosystem services in South Africa's Little Karoo: synergies and costs. Conservation Biology.

BE initiated the study after a workshop with all co-authors. BE did the analysis with support from JC and MB. BE wrote the chapter with support from BR. All co-authors provided input to the manuscript.

A key question when conducting and implementing conservation plans is: what can we conserve given a particular budget? This questions and new advances in conservation planning require the inclusion of cost data, budgets and estimates of the biodiversity benefit obtained from a given action or investment. This chapter explores these advances in the light of the integration of ecosystem services in conservation assessments that also take opportunity costs of conservation into consideration. It seeks to answer the question of how much more does it cost to safeguard ecosystem services in addition to biodiversity. This chapter includes elements of the valuation component of the framework (Figure 1.3). The discussion is centred around the benefits of planning for services and what effects this could have on the cost of implementation.

1.3.6 What have we learnt, what are the challenges and what is the way forward?

Chapter 7: General discussion and conclusions

The final chapter synthesises the previous chapters and attempts to distil the progress made in this thesis, the challenges encountered and some ideas on future research needs.

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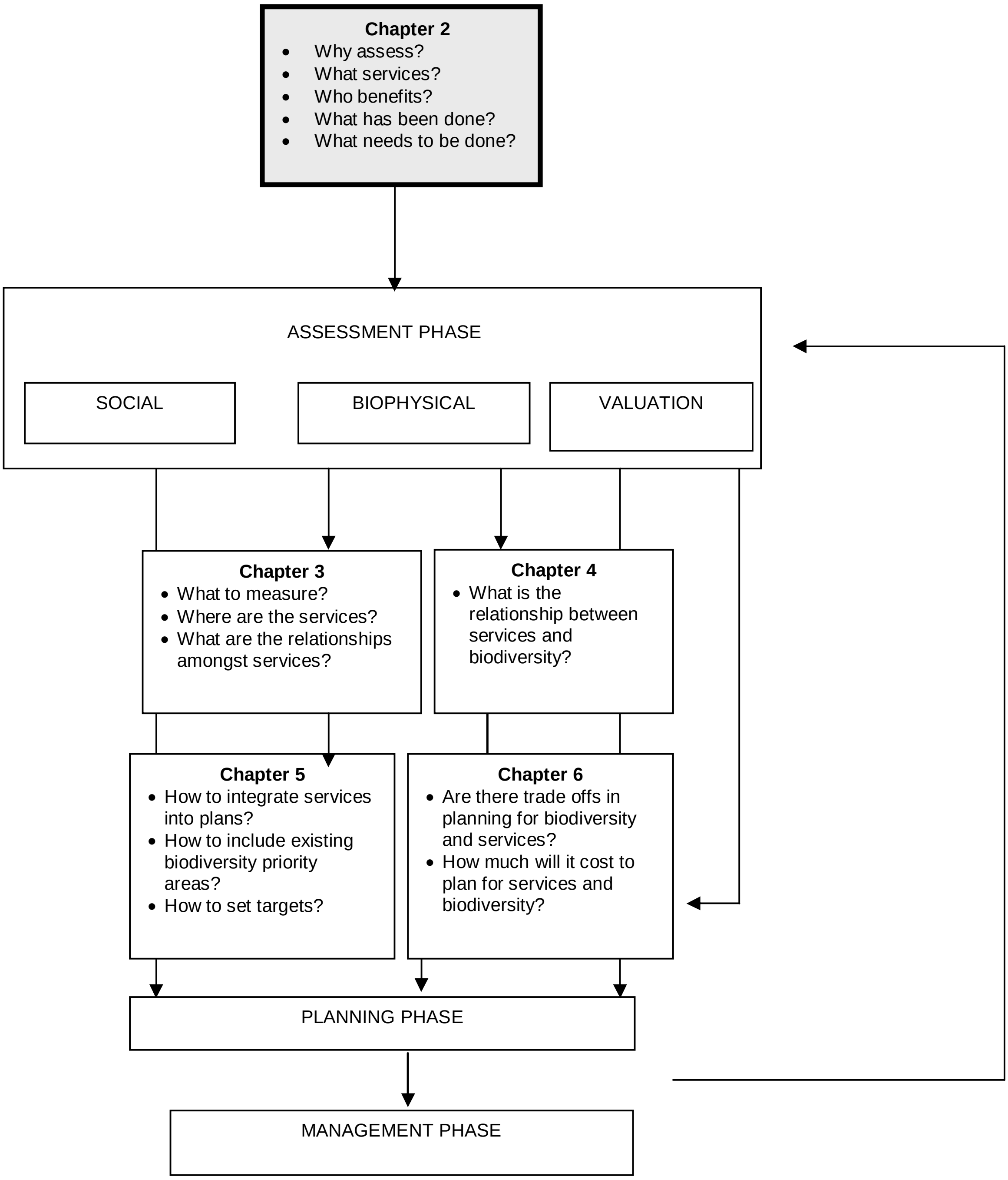
Chapter 2
Why assess?
What services?
Who benefits?
What has been done?
What needs to be done?
ASSESSMENT PHASE
SOCIAL
BIOPHYSICAL
VALUATION
Chapter 3
What to measure?
Where are the services?
What are the relationships amongst services?
Chapter 4
What is the relationship between services and biodiversity?
Chapter 5
How to integrate services into plans?
How to include existing biodiversity priority areas?
How to set targets?
Chapter 6
Are there trade offs in planning for biodiversity and services?
How much will it cost to plan for services and biodiversity?
PLANNING PHASE
MANAGEMENT PHASE

CHAPTER 2. INTEGRATING ECOSYSTEMS SERVICES INTO CONSERVATION ASSESSMENTS: A REVIEW

ABSTRACT

A call has been made for conservation planners to include ecosystem services into their assessments of conservation priority areas. The need to develop an integrated approach to meeting different conservation objectives and a shift in focus towards human wellbeing are some of the motivations behind this call. There is currently no widely accepted approach to planning for ecosystem services. This study contributes towards the development of this approach through a review of conservation assessments and the extent to which they include ecosystem services. Of the 476 conservation assessments identified by a set of search terms on the Web of Science, 100 were randomly selected for this review. Of these only seven had included ecosystem services, while another 13 had referred to ecosystem services as a rationale for conservation without including them in the assessment. The majority of assessments were based on biodiversity pattern data while 19 used data on ecological processes. A total of 11 of these 19 assessments used processes, which could be linked to services. Ecosystem services have witnessed an increase in attention received in conservation assessments since the year 2000, however trends were not apparent beyond this date. In order to assess which types of ecosystem services and how they have been accounted for in conservation assessments, I extended the review to include an additional nine conservation assessments which included ecosystem services. The majority included cultural ecosystem services, followed by regulatory, provisioning and supporting services respectively. I conclude with an analysis of the constraints and opportunities for the integration of ecosystem services into conservation assessments and highlight the urgent need for an appropriate framework for planning for ecosystem services.

Keywords: biodiversity conservation, ecosystem services, ecological processes, ecosystem functions, area selection, conservation planning

2.1 Introduction

Conservation planning is a rapidly evolving field whose goal is to minimise the loss of biodiversity through the selection of areas for conservation action (Pressey and Cowling, 2001). It is increasingly being used to locate or expand protected areas, to direct funding, and to influence land use decision-making (Margules and Pressey, 2000). I distinguish between conservation assessment and conservation planning as distinct activities, as the two are often conflated. Conservation assessment involves identifying spatial priorities for conservation action (i.e. area selection). When complemented with the development of an implementation strategy, in the context of stakeholder collaboration (i.e., the involvement of agencies who will take implementation of the plan forward), these activities constitute conservation planning (Knight et al., 2006a). Techniques for conservation assessment are evolving rapidly and new data sets are continually being introduced (Ferrier, 2002; Margules et al., 2002; Williams et al., 2002). Data commonly used in conservation assessments are species distribution records and broad-scale attributes obtained from data on species communities and/or abiotic data (e.g. land types; Lombard et al., 2003; Brooks et al., 2004). Recent conservation plans have also begun to include areas, which maintain ecological and evolutionary processes important for biodiversity persistence (e.g. interspecific interactions, regular and nomadic faunal movements, disturbance regimes; Balmford et al., 1998; Cowling et al., 1999; Rouget et al., 2003).

In addition to these data types, some authors have called for the inclusion of ecosystem services in conservation plans (Balvanera et al., 2001; Singh, 2002; Chan et al., 2006). Ecosystem services are the benefits that humans derive from ecosystems (e.g. fuel wood, water purification, recreation), which ultimately underpin human well-being (Millennium Ecosystem Assessment, 2003). The inclusion of ecosystem services into conservation assessments would allow for the development of an integrated approach to evaluating the merits and congruence of different conservation objectives (Balvanera et al., 2001; Singh, 2002). Furthermore, it would place a specific focus on safeguarding human well-being (Balvanera et al., 2001; Kremen and Ostfeld, 2005) which might contribute to improving the societal relevance of conservation assessments, which should better support their translation into effective conservation action (Knight et al., 2006a).

Currently, a widely endorsed methodology for planning for ecosystem services does not exist (Balvanera et al., 2001; Kremen, 2005). In order to contribute towards the development of such a methodology, this study conducts a survey of the peer-reviewed literature on conservation assessments. It assesses the ways and extent to which ecosystem services have been integrated into conservation assessments by evaluating three questions: 1) to what extent have ecosystem services been included as features in conservation assessments?; 2) what types of ecosystem services have been included?; and 3) how have these services been accounted for in conservation assessments? I conclude by discussing opportunities and constraints for incorporating ecosystem services into conservation assessments.

2.2 Methods

2.2.1 Scope of the study

Although the term ecosystem services has been defined numerous times (e.g. Daily, 1997; Costanza et al., 1997; Ekins, 2003; Millennium Ecosystem Assessment, 2003), these definitions are often competing and do not standardise the meaning, constraints and measurement of ecosystem services (Boyd and Banzhaf, 2006). In addressing the review's aims it is useful to define what I mean by ecosystem services, as well as the scope of this study. The terms ecosystem functions, ecological services, ecological functions, environmental services and environmental functions are sometimes used interchangeably with the term ecosystem services (e.g. de Groot, 1992; Costanza et al., 1997; Daily, 1997). I define ecosystem services as ecosystem functions that provide benefits to humans i.e. a human beneficiary (current or future) must be explicit. There is a lack of agreement on the source of ecosystem services with some authors stating that natural systems provide these services (Daily, 1997), while others say that they can be provided by human-modified and natural systems (Costanza et al., 1997; Millennium Ecosystem Assessment, 2003). This review, with its focus on conservation, is limited to services provided largely by natural systems. Marine ecosystems were excluded from this study because there have been comparatively few conservation assessments undertaken for the marine environment.

2.2.2 The review

This review made use of the Web of Science (http://www.newisiwebofknowledge.com) to search for English language, peer-reviewed publications published from 1998–2005. I chose

1998 as the start date for the study as the influential book “Nature's Services” (Daily, 1997) was published immediately prior to this year. I used the phrases “conservation assessments”, “conservation planning”, “conservation plan”, “conservation evaluation”, “conservation value”, “reserve selection”, “area selection”, “area identification”, “priority area”, “bioregional conservation”, “bioregional planning”, “ecoregional assessment”, “ecoregional conservation”, “integrated conservation” and “natural areas identification” commonly associated with conservation assessments to conduct the search. A total of 476 studies were identified and I randomly selected 100 of them for further analysis. I excluded a total of 12 studies from the sample because either they were conceptual papers or could not be located. I analysed the remaining 88 studies based on the first question to determine the extent to which they included ecosystem services.

Due to the small number of studies (seven) found to include ecosystem services, I could not address the remaining questions, dealing with the type of services and how they are accounted for in conservation assessments, from our original random sample of 88 papers. I therefore performed a further search on the remaining original 376 studies for papers that included services using the terms “ecosystem services”, “ecosystem function’, “environmental services”, “environmental function”, “ecological value”, “ecological services”, “ecological function” and “ecosystem value”. An additional nine conservation assessments were found, which have included ecosystem services in this second search making a total of 16 conservation assessments which was used to address these last questions. In order to evaluate which type of ecosystem services were included in these 16 conservation assessments, I subdivided ecosystem services into regulatory, supporting, provisioning and cultural services based on the MA (Millennium Ecosystem Assessment, 2003) conceptual framework. Finally, I reviewed the 16 conservation assessments in an attempt to classify the main ways in which ecosystem services have been included into conservation assessments.

2.3 Results and discussion

2.3.1 The extent of ecosystem service inclusion: Trends in conservation assessments

Although ecosystem services are frequently mentioned in conservation assessments, they have rarely been included. Of the 88 conservation assessments reviewed, 20 (23%) referred to ecosystem services as part of the rationale for conserving biodiversity. Of these only 7 (8%) included services. Ecosystem services did not feature at all in the remaining 68 (77%) conservation assessments that were considered. The review highlighted that biodiversity features such as species and land classes are still the most commonly used data in conservation assessments, occurring in 87 (99%) of the reviewed assessments (Figure 2.1). Data on ecological and evolutionary processes were used in 19 (22%) conservation assessments. Of the studies that included processes, 11 (59%) used surrogates for ecological processes that could be significant to service delivery, although these were not specifically targeted because of their importance to humans. This approach is discussed in more detail in Section 3.3. Figure 2.2 demonstrates an increasing awareness of ecosystem services and ecological processes in conservation assessments from the year 2000 onwards. This increased awareness could be due to the growing recognition of the importance of ecological and evolutionary processes, as well as ecosystem services in conservation biology (e.g. Daily, 1997; Balmford et al., 1998; Cowling et al., 1999). Trends in this awareness post 2000 are hard to identify based on the limited sample size.

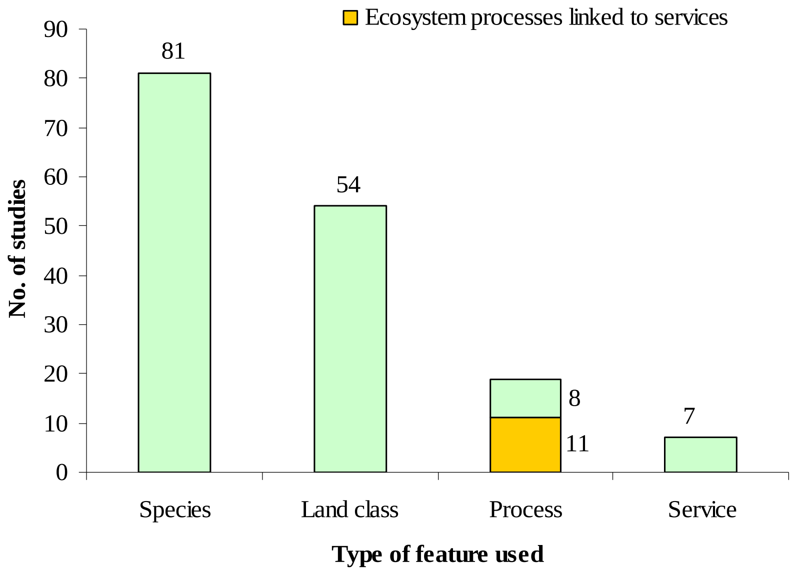

Figure 2.1: Types of data used in conservation assessments from 88 conservation assessments published between 1998 and 2005. Categories are not mutually exclusive as most conservation assessments use more than one type of feature data.

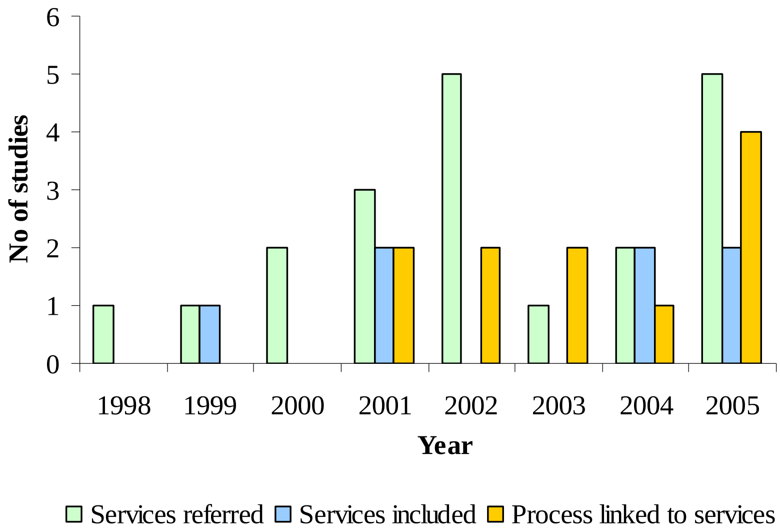

Figure 2.2: Results of a review of the integration of ecosystem services in 88 conservation assessments over time. Graph shows number of assessments that have either referred to or included services, reflecting the number of assessments which have included services and those that have included processes potentially linked to services.

2.3.2 What types of services have been included?

Our extended search for studies that have included ecosystem services in conservation assessments located a total of 16 assessments. Of these, the majority included cultural (10 (63%)), followed by regulatory services (8 (50%)), provisioning (7 (44%)) and supporting (2 (13%)) services (Table 2.1). These categories are not mutually exclusive; some assessments include multiple services. Aesthetic value received the most attention amongst cultural services. The main regulatory service included was water production. Because of the difficulties in distinguishing between water supply and water regulation, following de Groot et al. (2002) both services were treated as regulatory. Gas and climate regulation have also received some attention in conservation assessments either through the inclusion of ecosystem services or through the inclusion of ecological processes linked to these services.

2.3.3 How are ecosystem services accounted for in conservation assessments?

It became apparent from the review of the 16 conservation assessments that there are three main and overlapping ways in which ecosystem services have been integrated into conservation assessments: 1) through biodiversity pattern; 2) through ecological processes; and 3) through the mapping of services.

2.3.3.1. Through biodiversity pattern

Elements of biodiversity pattern (e.g. species and habitats) are important to humans and provide vital services such as the provision of medicines, fuel wood or building materials (Diaz et al., 2005; Millennium Ecosystem Assessment, 2005a). Higher levels in the biodiversity hierarchy (e.g. ecosystems; Noss, 1990) usually support ecological functions, including the processes that help in maintaining ecosystem viability and services (McKenzie et al., 1989; Williams et al., 2002). In many conservation assessments, it is often assumed that conserving biodiversity pattern will also conserve ecosystem services (see Balvanera et al., 2001; Singh, 2002). However, as Singh (2002) points out, there is limited evidence of spatial congruence between areas important to species conservation and those important to ecosystem services. Therefore, this review does not recognise conservation assessments, which include data on biodiversity pattern only as having included ecosystem services.

However, some assessments (e.g. Camm et al., 2002) suggest that species or habitats with a higher value to humans be assigned higher weights or targets in the assessment. For example, Nagendra (2001) scored areas based on their importance to economically useful plants in the study area by measuring the economic value of each habitat for marketable products such as grass, timber and non-timber products. Particular attention is often given to habitats like forest and wetlands (e.g. wetlands of international importance, Hoctor et al., 2000; Pérez-Arteaga et al., 2002). Focusing on “special” habitats in a conservation assessment ensures that the services provided by them, such as carbon sequestration and water regulation, continue to be delivered. Of the 16 conservation assessments that included ecosystem services, about half of them gave higher weight to biodiversity pattern features, which provide services to humans. These were considered as having included ecosystem services in this review.

Table 2.1: Ecosystem services captured in 16 conservation assessments (see Methods for details on how these were identified). The services are categorised according to schemes presented by the Millennium Ecosystem Assessment (MA 2003) and de Groote et al. (2002)

Ecosystem service	Method of identification	Source
Regulatory		
Carbon sequestration	NDVI (Normalised differential vegetation index)	Nagendra, 2001
	Unclear	Faith et al., 2001; Phua and Minowa, 2005
Water production	Catchments for down stream human use	Cowling et al., 2003
	Vegetation, soil and slope	Gou and Gan, 2002,
	Rivers, lakes, dams	Arriaza et al., 2004
	Aquifer recharge areas	Hoctor et al., 2000
Flood prevention	Environmental variables*	Phua and Minowa, 2005
Drought	Environmental variables*	Phua and Minowa, 2005
Erosion control	Slope	Kremen et al., 1999
Supporting		
Productive Soils	Soil types	Cantu et al., 2004
	Local knowledge	Nagendra, 2001
Provisioning		
Shell fish production	Harvesting areas	Hoctor et al., 2000
Economic value	Economically useful plants	Coppolillo et al., 2004; Polasky et al., 2005
	Timber, rice, grass, vegetation type	Nagendra, 2001
Forest Production	Eco forestry potential	Kremen et al., 1999; Faith et al., 2001; Faith and Walker, 2002
Cultural		
Aesthetic value	Environmental variables* and mixed forest	Bojórquez-Tapia et al., 2004; Natori et al., 2005
	Outstanding scenic features and landscape quality	Cowling et al., 2003
	Scenic rivers	Hoctor et al., 2000
	Landscape quality	Arriaze et al., 2004
	Photos and environmental variables*	Mendel and Kirkpatrick, 1999
Cultural value	Valued species	Coppolillo et al., 2004; Phua and Minowa 2005
Ecotourism	Accessibility, expert knowledge	Cowling et al., 2003; Kremen et al., 1999
Environmental education	Vegetation types	Bojórquez-Tapia et al., 2004
Recreation	Expert knowledge	Cowling et al., 2003; Kurttila and Timo, 2003

** Environmental variables refer to any combination of slope, elevation, rainfall, soil depth, geology and topology*

2.3.3.2. Through ecological processes

In a similar fashion to selected biodiversity pattern elements, ecological processes, which support the persistence of biodiversity pattern (e.g. pollination, climate change resilience, nutrient cycling, primary productivity and sediment transport; Ibisch et al., 1999; Fairbanks et al., 2001; Bassett and Edwards, 2003; Pressey et al., 2003; Rouget et al., 2004), can form links to ecosystem services (see de Groot et al., 2002; Diaz et al., 2005). Eleven of the 88 conservation assessments that were reviewed included ecological processes that could be linked to ecosystem services, although the authors did not specifically target them because of their importance to humans. These processes include pollination, predation, dispersal, nutrient cycling, primary productivity, accommodation of climate change impacts and sediment dynamics (Lozano et al., 2003; Pyke et al., 2005). A conservation assessment, which specifically targets conservation of crop pollinators, would be seen as including ecosystem services, while one that targets the ecological process of pollination (not specifically of a crop species) would be seen as including a process. Therefore these 11 assessments were not classified as including ecosystem services in this review. However the strong potential links between processes and services implies that this approach could prove useful in the development of frameworks for integrating ecosystem services into conservation plans.

2.3.3.3. Through mapping ecosystem services

A final approach, and one that uses components of the previous two approaches, involves the mapping of ecosystem services as a feature in conservation assessments. This is made possible by the identification and mapping of ecosystem service providers (populations, species, habitat types or functional units) responsible for the provision of services (Kremen, 2005). The method used to capture each service will depend on the service in question. During the mapping process, the functional contribution of each entity to the service can be assessed while the individual ecosystem service providers are mapped (Balvanera et al., 2005; Kremen, 2005; van Jaarsveld et al., 2005). Sometimes, mapping the ecosystem service provider (ESP) may rely on abiotic attributes as well as biotic attributes. Some examples of ESPs mapped in conservation assessments include: vegetation types, slope, scenic rivers, useful plants and leaf litter (Kremen, 2005). For example, Gou and Gan (2002) modelled water retention (important for water regulation and supply) in a watershed in China using vegetation, soil and slope.

The mapping of ESPs is one of the most explicit methods for including services in conservation assessments and contributes to our ecological understanding of ecosystems services, essential for the effective management of these services (Kremen and Ostfeld, 2005). However, many ESPs are difficult to study experimentally and it remains a challenge to develop a clear link between the ESPs, ecosystem functions measured in experiments and the services they provide (Costanza et al., 1997; Schläpfer et al., 1999; Schwartz et al., 2000).

Some conservation assessments move beyond the mapping of services, setting targets for them and selecting areas which achieve ecosystem service and biodiversity targets (Nagendra, 2001; Phua and Minowa, 2005; Natori et al., 2005). Others use them as a form of trade off analysis, mapping areas important to biodiversity and those important to ecosystem services and assessing the spatial congruence or conflict potential (Faith et al., 2001; Faith and Walker, 2002; Bojorquez- Tapia et al., 1995; Polasky et al., 2005). Some assessments map areas of ecosystem service supply and areas of demand in an effort to evaluate and manage the benefit flows of these services (Kremen et al., 1999; van Jaarsveld et al., 2005).

2.4 Conclusions

This review confirms that despite calls for developing methods to include ecosystem services into conservation assessments and planning processes (Balvanera et al., 2001; Singh, 2002; Leslie, 2005), only a small number of peer-reviewed conservation assessments have actually done so. Below I discuss some of the constraints and opportunities that exist for the inclusion of services into conservation assessments.

2.4.1 Constraints

2.4.1.1. Data and knowledge constraints

In all conservation assessments, biodiversity has to be measured and mapped; goals have to be set and methods for implementation determined (Margules et al., 2002). With the focus now shifting to ecosystem services, the data and knowledge gaps and demands are obvious and significant (Kremen, 2005). Including ecosystem services into conservation assessments will require a proper understanding of the ecology of the service, its conservation or management requirements and the benefits to humans both in space and time (Kremen, 2005; Millennium Ecosystem Assessment, 2005b; van Jaarsveld et al., 2005). Although the techniques for conservation planning are well advanced, definitions, data and tools for mapping ecosystem services, which locate sources, sinks and threats relevant to each service, are virtually non-existent (Balvanera et al., 2001).

2.4.1.2. The challenge of defining ecosystem services

In addition to the data and technical constraints around mapping services, this review highlights possibly more important challenges around defining and measuring ecosystem services. Mapping services is much more than just mapping the ecological function supporting that service. It requires the identification of beneficiaries, their location and use of the service to translate a function into a service. Whether these are current or potential future beneficiaries should also be clarified. The identification and mapping of beneficiaries and their use of services is a daunting task made more so by complex trade relations or service transfers. Some ecosystem services can be used far beyond where they are produced and new users can emerge. For example, the current momentum on payments for ecosystem service around the world (see http://www.katoombagroup.org) has seen new demand for certain services emerging.

Another challenge to ascertaining when a service is a service arises around the identification of systems that provide services. Many scientists state that only natural systems provide ecosystem services, however this assumption is not universal and many assessments e.g. Costanza et al. (1997) and the Millennium Ecosystem Assessment (MA, 2005a,b) include both natural and human modified systems as the sources of services. As Boyd and Banzhaf (2005) state: "As the term [ecosystem services] gets more and more use, there is a danger it will become a soft, generic label signifying everything, yet nothing." It is therefore essential that these be clarified and other uncertainties around ecosystem services before we embark on efforts to map and integrate them into conservation plans.

2.4.1.3. Time frames and limited congruence

As conservation assessments are often conducted over short time frames due to funding limitations (Knight et al., 2006b), planners are left with very little time to explore the mapping of services. Furthermore, conservation assessments are initiated to meet specific goals of stakeholders. If the aim is to ensure biodiversity persistence (especially for particular species), as is the case with most conservation assessments (Brooks et al., 2004; but see Cowling et al., 2004), there will be little or no incentive to include areas important for the conservation of ecosystem services.

Although the persistence of biodiversity and the sustainability of ecosystem services are largely dependent upon one another, the two are not wholly interchangeable (Williams and Araújo, 2002). Therefore planning for biodiversity and ecosystem services simultaneously may be difficult as areas important for ecosystem services might not always be important for biodiversity (Balvanera et al., 2001, Chan et al., 2006). For example the most scenic rivers (cultural service) may not be the ones containing irreplaceable biodiversity features (Singh, 2002). Although we know that many ecosystem services are ultimately dependent on biodiversity (Ostfeld and Logiudice, 2003; Millennium Ecosystem Assessment, 2005a), there is currently considerable debate over the extent to which measures of biodiversity adequately act as surrogates for ecosystem services (Lacroix and Abbadie, 1998; Balvanera et al., 2001; Cottingham et al., 2001; Swift et al., 2004), and hence the extent to which conserving biodiversity also conserves ecosystem services.

In addition to this lack of congruence, other factors impact on the feasibility of planning for biodiversity and ecosystem services simultaneously. These factors include differences in the systems under consideration, the agencies responsible and the implementation arrangements. Ecosystem services and biodiversity differ in that services can be provided by production landscapes (e.g. cultivated fields), which are usually not the focus of conservation assessments (although see Green et al., 2005). Biodiversity is largely the domain of conservation agencies, while ecosystem services often fall under broader resource management agencies. Therefore, the implementation strategies and tools for biodiversity (e.g. land acquisition) may differ from those for ecosystem services (e.g. resource management). In such cases, planning for ecosystem services might require different approaches to those employed in conservation assessments. These constraints are based on experience in the terrestrial environment and will be even more severe in the marine and freshwater environments where conservation plans and ecosystem services are less researched.

2.4.2 Opportunities

2.4.2.1. Tools and principles

Despite these constraints, I believe that the tools and principles of conservation assessments (Margules and Pressey, 2000), as well as the lessons learnt by conservation planners (Knight et al., 2006b) provide valuable insights and starting points for ecosystem service planning. For some services the data, techniques and software provided by conservation assessments already exist and are well suited for planning for ecosystem services. For example, van Jaarsveld et al. (2005) used an irreplaceability analysis (Ferrier et al., 2000) to generate maps of priority areas for ecosystem services. Software such as C-Plan (Pressey, 1999) and MARXAN (Ball and Possingham, 2000), which are underpinned by explicit targets for biodiversity features and which generate maps of irreplaceability, lend themselves easily to assessments for ecosystem services.

2.4.2.2. Available data

Many of the services included in conservation assessments were mapped from datasets that are commonly used in conservation assessments (e.g. vegetation types, aspect, slope in the case of scenery; Mendel and Kirkpatrick, 1999; Bojorquez-Tapia et al., 1995). In some cases, including ecosystem services may simply mean refining targets for some biodiversity features to capture, for example, species known to contribute to a particular service. Various approaches have also been developed for the measurement and mapping of ecosystem services in economic valuations. But data generated from economic valuations or other socio-economic data sets may not necessarily be suitable for planning for ecosystem services due to the level of uncertainty often associated with such data (Farber et al., 2002; Chee, 2004). However, economic valuations can provide a first step for the setting of targets for services. Although current techniques can be used to plan for some ecosystem services, new data and techniques will be required for the majority of ecosystem services.

2.4.2.3. Implementation focus

The recent shift in focus in conservation planning from assessment techniques to “planning for implementation” (Pierce et al., 2005; Knight et al., 2006a) provides increasing impetus for including ecosystem services into conservation assessments. Ecosystem services contribute to this implementation focus of conservation plans in a number of ways: 1) Payments for ecosystem services are potentially a strong avenue for securing priority areas. 2) Services have an advantage in that they are linked to beneficiaries and thus facilitate the implementation of conservation plans. One of the challenges of implementing outputs from conservation assessments is the fact that implementation success is dependent upon the choices people make (Freyfogle and Newton, 2002; Cowling and Pressey, 2003) and so should be guided by human values (Theobald et al., 2000). 3) Targeting services in conservation assessments may achieve many biodiversity targets under an easy-to-sell umbrella of ecosystem services while at the same time improving the relevance of conservation plans to human well being.

In summary, mapping and conserving ecosystem services is important for sustainable development and may also achieve many conservation goals where congruence exists between areas of high biodiversity value and areas important for service delivery. The extent to which ecosystem services can be included in conservation assessments remains largely untested. I strongly urge conservation biologists and ecologists to develop appropriate methods for mapping ecosystem services and their benefit flows.

Planning for biodiversity and ecosystem services however requires some caution. Not only is the extent of concordance between priority biodiversity features and the spatial features required for ecosystem service delivery largely unknown, biodiversity and ecosystem services are associated with different values (intrinsic vs. utilitarian) and require different stakeholders and agencies (conservation agencies vs. resource managers) for effective implementation of actions. I therefore highlight an urgent need to develop an appropriate conceptual framework, an operational model (e.g. Knight et al., 2006a) and software tools for planning for ecosystem services.

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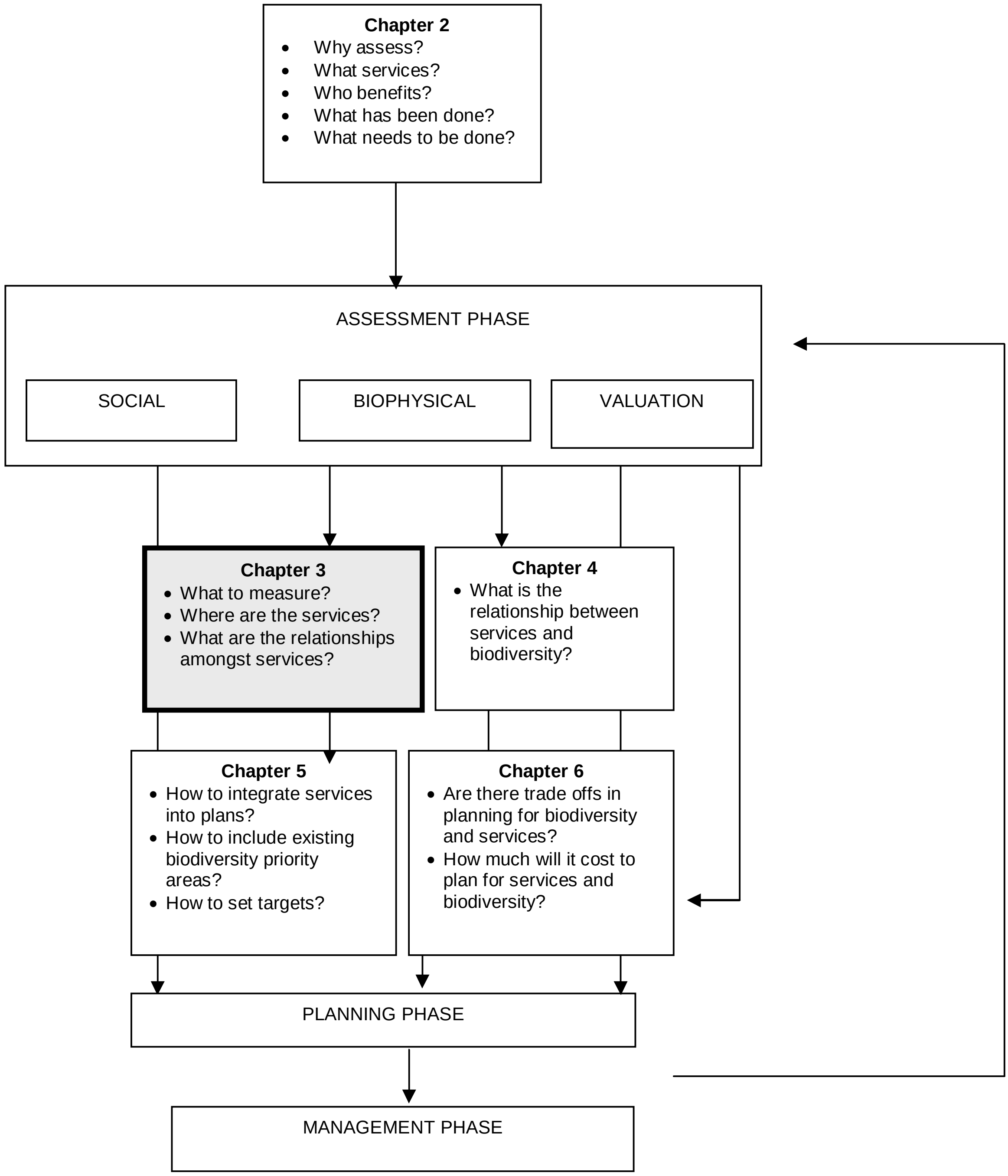
Chapter 2
Why assess?
What services?
Who benefits?
What has been done?
What needs to be done?
ASSESSMENT PHASE
SOCIAL
BIOPHYSICAL
VALUATION
Chapter 3
What to measure?
Where are the services?
What are the relationships amongst services?
Chapter 4
What is the relationship between services and biodiversity?
Chapter 5
How to integrate services into plans?
How to include existing biodiversity priority areas?
How to set targets?
Chapter 6
Are there trade offs in planning for biodiversity and services?
How much will it cost to plan for services and biodiversity?
PLANNING PHASE
MANAGEMENT PHASE

CHAPTER 3. MAPPING ECOSYSTEM SERVICES FOR PLANNING AND MANAGEMENT

ABSTRACT

This study mapped the production of five ecosystem services in South Africa: surface water supply, water flow regulation, soil accumulation, soil retention, and carbon storage. The relationship and spatial congruence between services were assessed. The congruence between primary production and these five services was tested to evaluate its value as a surrogate or proxy ecosystem service measure. This study illustrates that (1) most of South Africa's land surface is important for supplying at least one service, (2) there are low levels of congruence between the service ranges and even lower levels between the hotspots for different ecosystem services, and (3) primary production appears to show some potential as a surrogate for ecosystem service distribution. The implications of a heterogeneous landscape for the provision of ecosystem services and their management are highlighted and the potential for managing such services in a country like South Africa is discussed.

Keywords: Conservation biogeography, conservation planning, water, soil, carbon, primary productivity

3.1 Introduction

Several studies have reported a widespread decline in and unsustainable use of ecosystem services across the world (WRI, 2001; MEA, 2005). Areas that are important for maintaining ecosystem components and functions that provide ecosystem services have to be carefully managed to secure the provision of ecosystem services presently and in future (van Jaarsveld et al., 2005; Chan et al., 2006). Cowling et al. (2008) proposed an operational framework for mainstreaming the management of ecosystem services into all resource management sectors. This framework highlights the need to combine assessments of the biophysical, economic and social context with considerations of implementation opportunities and constraints into strategy development, implementation and management involving stakeholders. This study addresses the biophysical assessment which is similar in some respects to the assessment phase of systematic conservation planning which deals with the identification of geographic areas to ensure the effective conservation of biodiversity.

The aims of this study were to develop national-scale maps of selected ecosystem services in South Africa to inform and direct agencies responsible for their management, to develop methods for mapping these ecosystem services, and to evaluate the relationships between these services in order to assess whether a particular service can act as an umbrella or surrogate for other services. In addressing these aims, this study borrowed extensively from the field of conservation planning and the lessons learnt from conservation biogeography, which have proved useful in mapping biodiversity pattern and process for use in spatially-explicit decision making (Whittaker et al., 2005).

Conservation planning has come to rely heavily on the notion of surrogates (van Jaarsveld et al., 1998). Surrogacy is a relationship between an ‘‘indicator’’ parameter and an ‘‘objective’’ parameter (sometimes called a ‘‘target’’ parameter, what we ultimately hope to conserve) (Sarkar et al., 2005). For example, conservation biologists often use well studied taxa as surrogates for poorly studied groups. This study relies on these ideas of surrogacy, where ecosystem components and functions are often used as the surrogate or proxy for mapping the distribution of an ecosystem service. The notion of surrogacy between services (and primary productivity) is also tested to examine the potential for umbrella services.

South Africa is an appropriate place to test these ideas on ecosystem services and surrogacy. The country has been the focus of much conservation planning and biodiversity surrogate research (van Jaarsveld et al., 1998; Cowling et al., 2003), as well as agricultural and hydrological research (Schulze, 1997; Schoeman et al., 2002). Many good biophysical databases and assessment techniques are available (Balmford, 2003). In addition, the Southern African Millennium Ecosystem Assessment included South Africa in its analyses of ecosystem services (Biggs et al., 2004; van Jaarsveld et al., 2005), initiating the collation of new databases on ecosystem services.

3.2 Method

The study area, South Africa, covers approximately 1.22 million km^2. The country is classified as semi-arid and the rainfall varies geographically from less than 50 to about 3000 mm per year (annual mean 450 mm). The low and uneven distribution of rainfall coupled with very few perennial rivers (most of them shared with other countries) makes South Africa a water scarce country (Biggs et al., 2004). The soils are mostly very shallow with limited irrigation potential (Laker, 2005). They are extremely vulnerable to various forms of degradation (e.g. soil erosion, crusting and loss of organic matter) and have low resilience (Mills and Fey, 2004; Laker, 2005). Agriculture in South Africa is highly industrialised and food production relies largely on irrigation; 50% of the total water consumption is for irrigation and demand is increasing (Ministry of Agriculture and Land Affairs, 1998).

South Africa has a large human population of about 47 million (Stats SA, 2005). A large proportion of the population lives in rural areas on private commercial farms (1.5 million households) and the communal lands of the former homelands (2.3 million households) (van Horen and Eberhard, 1995). Most of the rural people survive as subsistence farmers and about 90% of the country's food consumption is met by domestic production with an ever increasing demand (Shackleton et al., 2001). Poverty alleviation is a national priority and initiatives such as payments for ecosystem services are being explored. Payments for ecosystem services are part of the emerging environmental investment sector in South Africa and are based on the commoditization of ecosystem services such as water supply and carbon storage (Peace Parks Foundation, 2005). Some of the country's semi-arid ecosystems store exceptionally large amounts of carbon (Mills et al., 2005).

3.2.1 Mapping ecosystem services

This study selected a suite of nationally relevant ecosystem services from de Groot et al. (2002)'s list of 23 ecosystem services. These include: surface water supply, water flow regulation, carbon storage, soil retention and accumulation. Surface water supply was separated from water flow regulation because of the complexity of the water provision service and the need to capture different components of the functions that are part of this service.

During mapping, ranges and hotspots were distinguished as has been done in spatial biodiversity assessments. The range of ecosystem services was defined as areas of meaningful supply, similar to a species' range or area of occupancy. The term ''hotspots'' was proposed by Norman Myers in the 1980s and refers to areas of high species richness, endemism and/or threat and has been widely used to prioritise areas for biodiversity conservation. Similarly, this study suggests that hotspots for ecosystem services are areas of critical management importance for the service. Here the term ecosystem service hotspot is used to refer to areas which provide large proportions of a particular service, and do not include measures of threat or endemism.

Several data sets were combined to produce a map for a service. Here thresholds were set for both the range and hotspot for each dataset and combined, implying that all areas within the ecosystem service range lie within the ranges of underlying layers. In some cases this integration of datasets had already been done prior to the study and thresholds for the single integrated layer were used.

3.2.1.1. Surface water supply

A number of previous ecosystem service studies have used water production, i.e. the volume of water produced by area, as an ecosystem service or as a surrogate for an ecosystem service (e.g. van Jaarsveld et al., 2005; Chan et al., 2006). Although the amount of water is an important benefit, it is not necessarily an ecosystem service on its own. The amount and distribution of rainfall is the primary determinant of the amount of water produced from a watershed. Rainfall patterns, in turn, depend mainly on abiotic factors such as regional climate systems and topography and not the ecosystems per se. Where the ecosystem does play a key role is in stabilising soils and filtering pollutants (e.g. fertilisers and pesticides), and thus regulating the water quality-the filtering service of de Groot et al. (2002). The total benefit to people of water supply is a function of both the quantity and quality with the ecosystem playing a key role in the latter. However, due to the lack of suitable national scale data on water quality for quantifying the service, runoff was used as an estimate of the benefit where runoff is the total water yield from a watershed including surface and subsurface flow. This assumes that runoff is positively correlated with quality, which is the case in South Africa (Allanson et al., 1990). Most of the country's surface water is generated in a few areas with high runoff: 50% of the runoff is generated by catchments comprising only 12% of the total area. Management of these areas will maintain or improve water quality because when they are kept in a good condition they yield high quality water, with the lowest possible soil erosion, nutrient and sediment loss (Scanlon et al., 2007).

In South Africa, water resources are mapped in water management areas called catchments (watersheds) where a catchment is defined as the area of land that is drained by a single river system, including its tributaries (DWAF, 2004). There are 1946 quaternary (4th order) catchments in South Africa, the smallest is 4800 ha and the average size is 65,000 ha. Schulze (1997) modelled annual runoff for each quaternary catchment. During modelling of runoff, he used rainfall data collected over a period of more than 30 years, as well as data on other climatic factors, soil characteristics and grassland as the land cover. In this study, median annual simulated runoff was used as a measure of surface water supply. The volume of runoff per quaternary catchment was calculated for surface water supply. The range (areas with runoff of 30 million M^3 or more) and hotspots (areas with runoff of 70 million m^3 or more) were defined using a combination of statistics and expert inputs due to a lack of published thresholds in the literature.

3.2.1.2. Water flow regulation

Water flow regulation is a function of the storage and retention components of the water supply service (de Groot et al., 2002). The ability of a catchment to regulate flows is directly related to the volume of water that is retained or stored in the soil and underlying aquifers as moisture or groundwater; and the infiltration rate of water which replenishes the stored water (Kittredge, 1948; Farvolden, 1963). Groundwater contribution to surface runoff is the most direct measure of the water regulation function of a catchment.

Data on the percentage contribution of groundwater to baseflows were obtained from DWAF (2005) per quaternary catchment and expressed as a percentage of total surface runoff, the range and hotspot being defined as areas with at least 10% and 30%, respectively (Colvin et al., 2007).

3.2.1.3. Soil retention

Areas where vegetation cover retains soils need to be managed carefully to allow for the continuous delivery of the services supporting land productivity and preventing damage from erosion through sedimentation and eutrophication of nearby rivers. Soil erosion removes nutrients and reduces fertility (DeFries et al., 2004). In South Africa large productive grazing areas have been lost through soil erosion (Kakembo and Rowntree, 2003).

Soil retention was modelled as a function of vegetation or litter cover and soil erosion potential. Schoeman et al. (2002) modelled soil erosion potential and derived eight erosion classes, ranging from low to severe erosion potential for South Africa. The vegetation cover was mapped by ranking vegetation types using expert knowledge of their ability to curb erosion. I used Schulze (2004) index of litter cover which estimates the soil surface covered by litter based on observations in a range of grasslands, woodlands and natural forests. According to Quinton et al. (1997) and Fowler and Rockstrom (2001) soil erosion is slightly reduced with about 30%, significantly reduced with about 70% vegetation cover. The range of soil retention was mapped by selecting all areas that had vegetation or litter cover of more than 30% for both the expert classified vegetation types and litter accumulation index within areas with moderate to severe erosion potential. The hotspot was mapped as areas with severe erosion potential and vegetation/litter cover of at least 70% where maintaining the cover is essential to prevent erosion. An assumption was made that the potential for this service is relatively low in areas with little natural vegetation or litter cover.

3.2.1.4. Soil accumulation

Soil scientists often use soil depth to model soil production potential (soil formation) (Heimsath et al., 1997; Yuan et al., 2006). The accumulation of soil organic matter is an important process of soil formation which can be badly affected by habitat degradation and transformation (de Groot et al., 2002). Soil depth and leaf litter were used as proxies for soil accumulation. Soil depth is positively correlated with soil organic matter (Yuan et al., 2006);

deep soils have the capacity to hold more nutrients. Litter cover was described above. Data on soil depth were obtained from the land capability map of South Africa and thresholds were based on the literature (Schoeman et al., 2002; Tekle, 2004). Areas with at least 0.4 m depth and 30% litter cover were mapped as important areas for soil accumulation, i.e. its geographic range. The hotspot was mapped as areas with at least 0.8 m depth and a 70% litter cover.

3.2.1.5. Carbon storage

In this study, only carbon storage was mapped because of a lack of data on the other functions related to the regulation of global climate such as carbon sequestration and the effects of changes in albedo. Carbon is stored above or below the ground and South African studies have found higher levels of carbon storage in thicket than in savanna, grassland and renosterveld (Mills et al., 2005). This information was used by experts to classify vegetation types (Mucina and Rutherford, 2006), according to their carbon storage potential, into three categories: low to none (e.g. desert), medium (e.g. grassland), high (e.g. thicket, forest) (Rouget et al., 2004). All vegetation types with medium and high carbon storage potential were identified as the range of carbon storage. Areas of high carbon storage potential where it is essential to retain this store were mapped as the carbon storage hotspot.

3.2.2 Evaluating ecosystem service congruence

The coarsest resolution in the data sets was the scale of catchments (watersheds) and thus all other data (soil accumulation, soil retention and carbon storage) were converted to this resolution. The delivery of each of the services was summarised as the median per catchment and relationships between services were evaluated using Spearman’s rank correlations. Primary productivity per catchment from Schulze (1997) was included in the correlation analysis because primary productivity is believed to be a good surrogate for ecosystem function (Tilman et al., 1997). Spatial overlap between services was calculated using proportional overlap (Prendergast et al., 1993) which expresses the area shared between two services as a percentage of the area of the service with a smaller total area. Service richness was expressed as the number of service ranges, as well as hotspots, per catchment. Only catchments where the range or hotspot of a service covered more than an arbitrary yet inclusive threshold of 10% of the catchment were included in the richness analysis.

3.3 Results

Soil accumulation had the largest range, covering about 43% of South Africa (Figure 3.1). Soil retention also had a relatively large range of about one third of the country, followed by water flow regulation (28%) and carbon storage (26%). Surface water supply showed the smallest range of 21%. The carbon storage hotspot had the smallest area (3%) of all hotspots and soil accumulation the largest (14%).

The maps of service richness (Figures 3.2 and 3.3) mirrored the distribution of the two water services and carbon storage. Interestingly, the areas that were most important for surface water supply differed from those important for water flow regulation. Surface water supply was highest in the east, while areas important for water flow regulation were mostly in the central and northern parts of the country. This was partly because the high supply areas were found mainly in eastern montane areas with shallow soils over bedrock and little storage. Carbon storage was greatest in the eastern and northern areas. Unlike the water and carbon services which clustered in the same areas, the soil services were evenly distributed across the country, except in the south west.

Ninety-four percent of catchments in South Africa delivered at least one service, but few catchments produced more than three services. Only 5% of the catchments produced all five services when the ranges were considered and none were a hotspot for all five services (Figures 3.2 and 3.3). Some 30% of all catchments, mostly in the northern, eastern and southern parts of the country, produced at least four services, but only 7% were hotspots for four services, all of which were situated in the north and east.

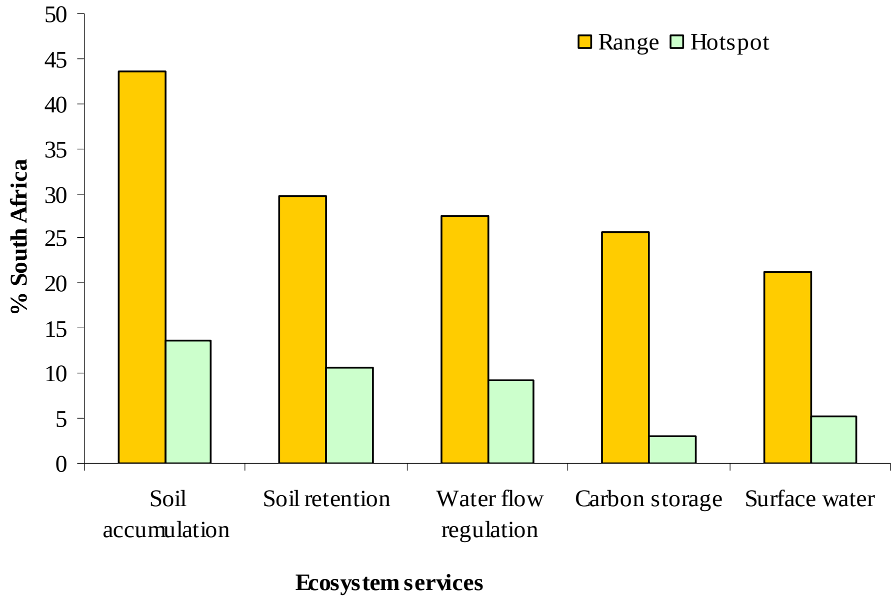

Figure 3.1: Percentage of South Africa that is important for the delivery of various ecosystem services, based on the geographical ranges and hotspots of the services. Both natural and transformed areas were included.

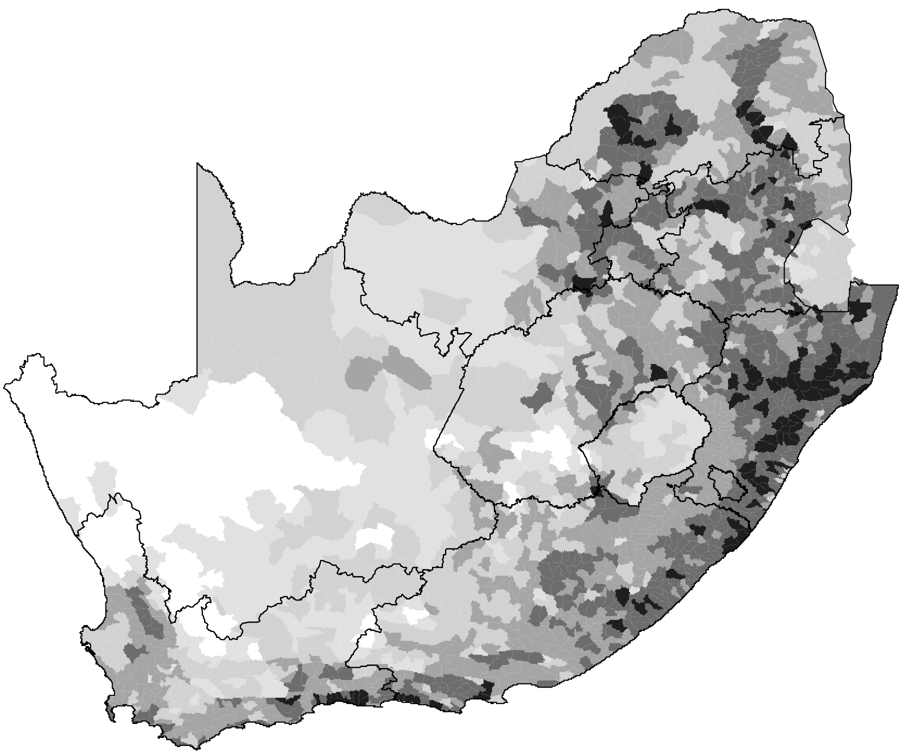

Figure 3.2: The number of ecosystem service ranges per catchment. The hotspot of a service is nested within its range.

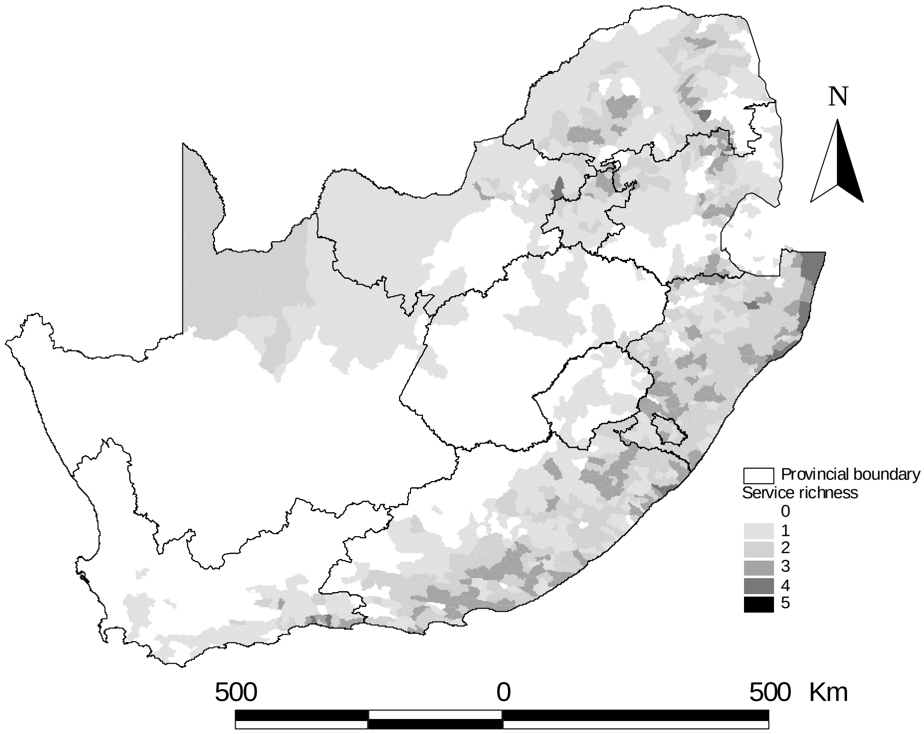

Figure 3.3: The number of ecosystem service hotspots per catchment.

3.3.1 Relationships between services

3.3.1.1. Ranges

Correlations between service ranges were generally low with more positive correlations than negative ones (Table 3.1). Soil retention showed a relatively strong positive correlation with other services, especially with soil accumulation and surface water supply. Correlation between surface water supply and water flow regulation was negative. Surprisingly, carbon storage showed a very weak and mostly negative correlation with other services and primary productivity. However, correlations between primary production and the other services were moderately strong and mostly positive. The three services that showed a relatively strong positive correlation with each other (surface water supply, soil accumulation and retention), were also positively correlated with primary productivity. Primary productivity was highly correlated ($r = 0.5$, $p < 0.05$) with ecosystem service richness (number of services per catchment). Despite the generally weak correlations between services, spatial overlap of areas providing services was relatively high (>30%; Table 3.2). Soil accumulation had the highest spatial overlap of more than 45% with all other services.

3.3.1.2. Hotspots

The correlations between ecosystem service hotspots were generally weaker than between service ranges (Table 3.1). Again soil retention showed the strongest positive correlation with soil accumulation. Despite the relatively high correlation between the ranges of soil retention and surface water supply, the correlation between the hotspots was not significant. Correlations between primary productivity and hotspots of services were weaker than those for the ranges. The correlation between primary productivity and the number of services per catchment was also high for hotspots ($r = 0.5$, $p < 0.05$). Spatial overlap between services was much lower for hotspots than for ranges (Table 3.2).

Table 3.1: Correlation values for ecosystem service (hotspots are shown in brackets). Only significant correlations are reported (p<0.05).

	Surface water supply	**Water flow regulation**	**Soil accumulation**	**Soil retention**	**Carbon storage**
Surface water	x				
Water flow regulation	-0.08 (**-0.09**)	x			
Soil accumulation	0.14 **(-0.15)**	0.14 **(0.14)**	x		
Soil retention	0.23	0.14 **(-0.08)**	0.56 **(0.44)**	x	
Carbon storage	0.08	-0.05	-0.14	-0.17	x
Primary productivity	0.5 **(0.25)**	0.31 **(0.14)**	0.44 **(-0.2)**	0.65 **(0.24)**	-0.14

Table 3.2: Proportional overlap of ecosystem services ranges and hotspots (hotspots are shown in brackets).

	Water flow regulation	**Surface water supply**	**Soil accumulation**	**Soil retention**
Surface water	38.7 (**7.1**)	X		
Soil accumulation	55.9 (**12.9**)	52.5 (**20.7**)	X	
Soil retention	33.6 (**7.7**)	37.8 (**8.8**)	47.1 (**40.3**)	X
Carbon	39.7 (**7.5**)	27.2 (**4.4**)	52.3 (**19.8**)	28.9 (**23.9**)

3.4 Discussion

Although the ranges of most of the ecosystem services occupy less than one third of the country (Figure 3.1), the low levels of congruence suggest that almost all of the country is important for supplying at least one service with few areas supplying more than three services (Table 3.2, Figure 3.2). The heterogeneity of South Africa's landscape and its ecosystem services has important consequences for their management. Although this study only mapped five ecosystem services, these results imply that management for these and other services will be a resource and land intensive task, with little hope of focusing efforts on small areas which deliver multiple ecosystem services. This aligns with findings on the distribution of biodiversity where 50% of the country is important for conserving South Africa's diversity of species and ecosystems (Reyers et al., 2007) and carries with it the same dubious distinction and large responsibility that the richness of South Africa's biodiversity carries for conservation biologists (Cowling et al., 1989).

Ecosystem service hotspots in this study are comparable to biodiversity hotspots based on species richness (see Reid, 1998). By focusing on ecosystem service hotspots, managers could potentially reduce the resources and effort required. This has been the case in South Africa where mountain catchments were set aside as protected areas for water production (Rouget et al., 2003). However, these hotspots are open to the same criticisms levelled at biodiversity hotspots, in that they do not necessarily achieve the goals of conservation or management because they are neither systematic nor based on a goal or objective (Margules and Pressey, 2000). The even lower levels of congruence and correlation between hotspots (Tables 3.1 and 3.2) parallel the findings of studies on biodiversity hotspots (Williams et al., 1996), and support calls to develop systematic approaches for planning for ecosystem services (Cowling et al., 2008), rather than relying on a scoring or hotspots approach.

The weak correlations between ecosystem services assessed in this study and in Chan et al. (2006) demonstrate that one cannot use one ecosystem service to plan for others. This agrees with findings in conservation biology where support for biodiversity surrogates is varied, and most authors recommend using all available data (Lombard et al., 2003 but see Sarkar et al., 2005). Although services do not appear to act as surrogates for other services, findings of a correlation between most of the services and primary productivity in this study, offers some hope for the use of primary productivity as a surrogate for ecosystem function and services, especially in areas where no service data are available at appropriate planning scales. The factors driving primary productivity are important drivers for many services, hence the observed pattern. Carbon storage does not appear to show this relationship. This is possibly due to the scale and expert-opinion based nature of our map of carbon storage. However, the extraordinary carbon storage potential found in low production areas in the east of the country (Mills et al., 2005) is an indication that production and accumulation are not necessarily driven by the same factors. Correlation between primary productivity and ecosystem services only applies anyhow to our suite of ecosystem services at a broad scale, and weakly to some of them, suggesting caution in the use of primary productivity as an ecosystem service surrogate.

3.5 Literature cited

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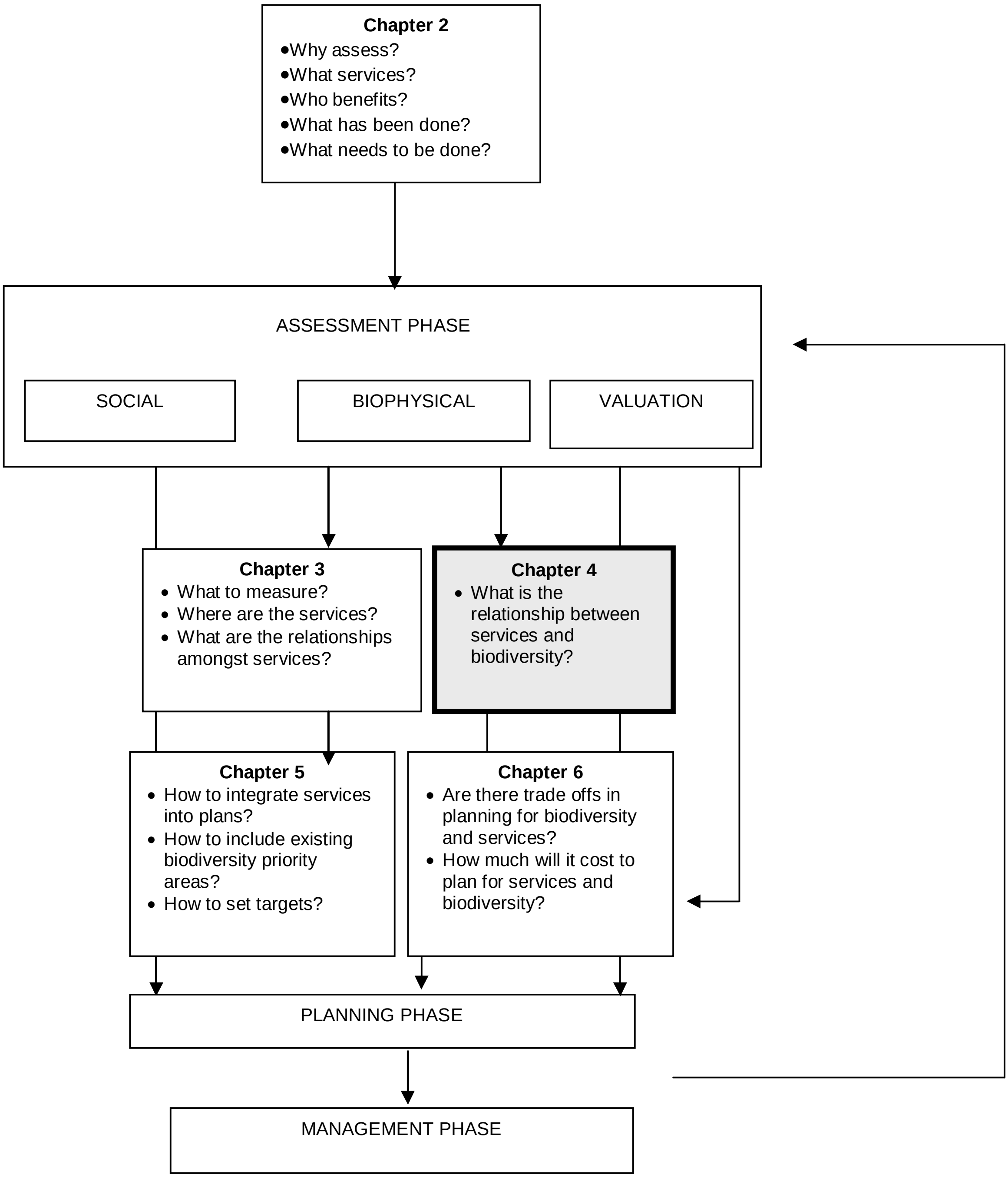

Chapter 2
•Why assess?
•What services?
•Who benefits?
•What has been done?
•What needs to be done?
ASSESSMENT PHASE
SOCIAL
BIOPHYSICAL
VALUATION
Chapter 3
What to measure?
Where are the services?
What are the relationships amongst services?
Chapter 4
What is the relationship between services and biodiversity?
Chapter 5
How to integrate services into plans?
How to include existing biodiversity priority areas?
How to set targets?
Chapter 6
Are there trade offs in planning for biodiversity and services?
How much will it cost to plan for services and biodiversity?
PLANNING PHASE
MANAGEMENT PHASE

CHAPTER 4. SPATIAL CONGRUENCE BETWEEN BIODIVERSITY AND ECOSYSTEM SERVICES IN SOUTH AFRICA

ABSTRACT

Ecosystems services sustain humans all over the world. The unsustainable use of ecosystem services around the world has led to widespread degradation which now threatens human's health and livelihoods. Although the maintenance of ecosystem services is often used to justify biodiversity conservation actions, it is still unclear how ecosystem services relate to different aspects of biodiversity and to what extent the conservation of biodiversity will ensure the provision of services. The aim of this study was to find out whether biodiversity priorities, biomes, species richness and vegetation diversity hotspots co-occur in space with ecosystem services. The distribution of the ranges and hotspots of five ecosystem services (surface water supply, water flow regulation, carbon storage, soil accumulation and soil retention) was assessed in South African biomes. Coincidence, overlap and correlation analyses were used to assess spatial congruence between ecosystem services and species richness (plants and animals) and vegetation diversity hotspots. The grassland and savanna biomes contained significant amounts of all five ecosystem services. There was moderate overlap and a generally positive but low correlation between ecosystem services hotspots and species richness and vegetation diversity hotspots. Species richness was mostly higher in the hotspots of water flow regulation and soil accumulation than would be expected by chance. The water services showed different levels of congruence with species richness hotspots and vegetation diversity hotspot. These results indicate that actions taken to conserve biodiversity in South Africa will also protect certain ecosystem services and ecosystem services can be used to strengthen biodiversity conservation in some instances.

Keywords: conservation planning, species, hotspot, soil, water, carbon.

4.1 Introduction

Ecosystems services sustain humans all over the world and directly support more than one billion people in the world living in extreme poverty (Costanza et al., 1997; MA, 2005; World Bank, 2006; Turner et al., 2007). Ecosystem services are the benefits that humans derive from ecosystems and include provisioning (e.g. medicinal plants and firewood), regulatory (e.g. water purification and regulation), supporting (e.g. soil retention and formation) and cultural services (e.g. the use of nature for spiritual purposes; MA, 2003; de Groot et al., 2002). Biodiversity and ecosystem services are intrinsically linked: the former underpins most ecosystem services and the maintenance of ecosystem services is often used to justify biodiversity conservation actions because of its importance in sustaining human livelihoods (Bookbinder et al., 1998; MA, 2005; Naughton-Treves et al., 2005). The degradation and unsustainable use of ecosystems and its services around the world now threatens the health and livelihoods of many people (WRI, 2001; MA, 2005).

Despite the wide use of ecosystem services to argue for biodiversity conservation, it is unclear exactly how different aspects of biodiversity relate to ecosystem services, and to what extent conserving biodiversity will ensure the provision of these services. Conserving biodiversity and ecosystem services might require different strategies. Many provisioning and cultural services are provided directly by biodiversity components such as species, vegetation types and landscapes. For example, forest ecosystems provide food, firewood, fencing material and medicinal plants, amongst others, to communities (Myers, 1988; Naidoo and Ricketts, 2006). However, some ecosystems services (especially supporting and regulatory ones) are a function of many ecosystem properties and may not necessarily be conserved simply through conservation strategies geared towards any particular facet of biodiversity. For example, the maintenance of water regulatory services is provided by a combination of biotic and abiotic factors (Gou et al., 2001; Le Maitre et al., 2007) which require a landscape and land-use management approach across the entire watershed. At present, much conservation effort is geared toward biodiversity per se and knowledge on conserving ecosystem services is still in its infancy (Balvanera et al., 2001).

A widely used strategy for biodiversity conservation is the identification of spatial priorities where conservation efforts should be focused (Margules and Pressey, 2000). These priorities can be identified using different measures of biodiversity including species richness or endemism, vegetation diversity and biomes. Some progress has been made in recent decades in mapping ecosystem services (Costanza et al., 1997; Naidoo and Ricketts, 2006; Troy and Wilson, 2006; Egoh, et al., 2008). Where these biodiversity components or priorities overlap with areas important for the delivery of ecosystem services, conservation strategies aimed at biodiversity may safeguard ecosystem services. Conversely, in areas of overlap, ecosystem services can be used as additional justification for biodiversity conservation. However, to date few studies have evaluated the spatial concordance between biodiversity and ecosystem services because mapping the location of the services is still difficult. Chan et al. (2006) investigated the relationship between biodiversity and ecosystem services in California. They found a generally low correlation between biodiversity and ecosystem services and moderate overlap between the two. In another study, Turner et al. (2007) found a generally high overlap between biodiversity priorities and ecosystem services but their results were not consistent across all regions. The ambiguity of these findings suggests that there is a need to extend the investigation to other parts of the world.

South Africa represents an excellent opportunity for exploring the spatial relationship between ecosystem services and biodiversity. Its heterogeneous landscapes are extraordinarily rich in biodiversity (Cowling et al., 1989, 1997), and extensive research has identified and located effective biodiversity surrogates, and areas of importance to biodiversity conservation (van Jaarsveld et al., 1998; Reyers et al., 2001; Cowling et al., 2003; Lombard et al., 2003). Many recent conservation initiatives in South Africa are geared towards sustainability and poverty alleviation presenting an opportunity for safeguarding ecosystem services. For example, the Working for Water project is aimed at improving biodiversity and water services by removing invasive alien plants, but also seeks to create jobs for poor, unemployed people (van Wilgen et al., 1998).

The first National Spatial Biodiversity Assessment (NSBA) for South Africa has recently been completed (Reyers et al., 2007). The NSBA identified national biodiversity priorities and was part of the National Biodiversity Strategy and Action Plan (NBSAP) which aimed to direct the conservation and sustainable use of South Africa’s biodiversity, as part of the nation’s obligation as a signatory to the Convention on Biological Diversity (CBD). The revised NSBA (due in 2009) will include ecosystem services more explicitly. The importance of understanding the spatial relationship between ecosystem services and biodiversity can no longer be overlooked. In chapter two of this thesis, I mapped the "range" (areas where a particular service is produced in meaningful quantities) and "hotspots" (areas which provide large components of a particular service) of five ecosystem services in South Africa (Figure 4.1). In this study I use the five ecosystem services mapped and biodiversity data from the NSBA to assess the spatial congruence between ecosystem services and biodiversity in South Africa. I do this by asking four questions. First, how are ecosystem service ranges and hotspots distributed across biomes in South Africa? Second, to what extent do species hotspots overlap with ecosystem service hotspots? Third, is species richness higher than expected in ecosystem service hotspots? Fourth, do areas that are prioritised for biodiversity conservation overlap with ecosystem service hotspots?

a)

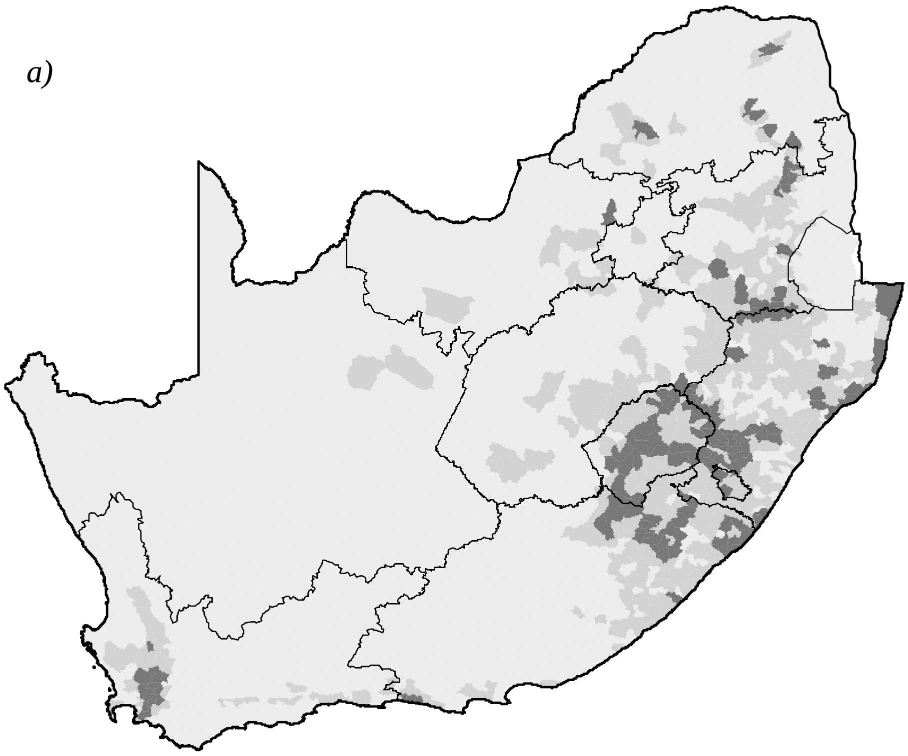
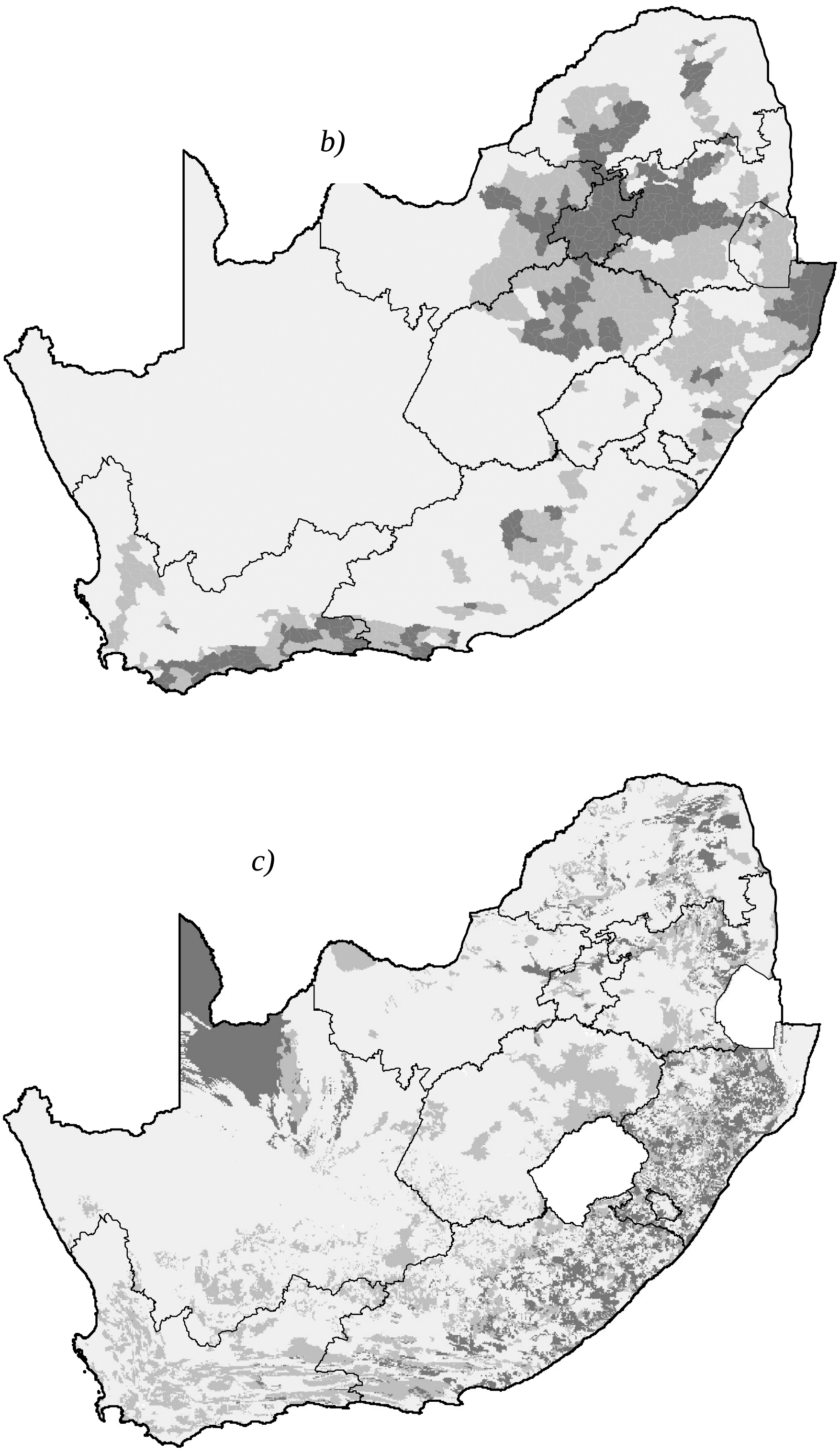


Figure 4.1. The extent of the range and hotspots of ecosystem services in South Africa used in this study. A) Surface water supply B) water flow regulation C) soil retention D) soil accumulation E) carbon storage.



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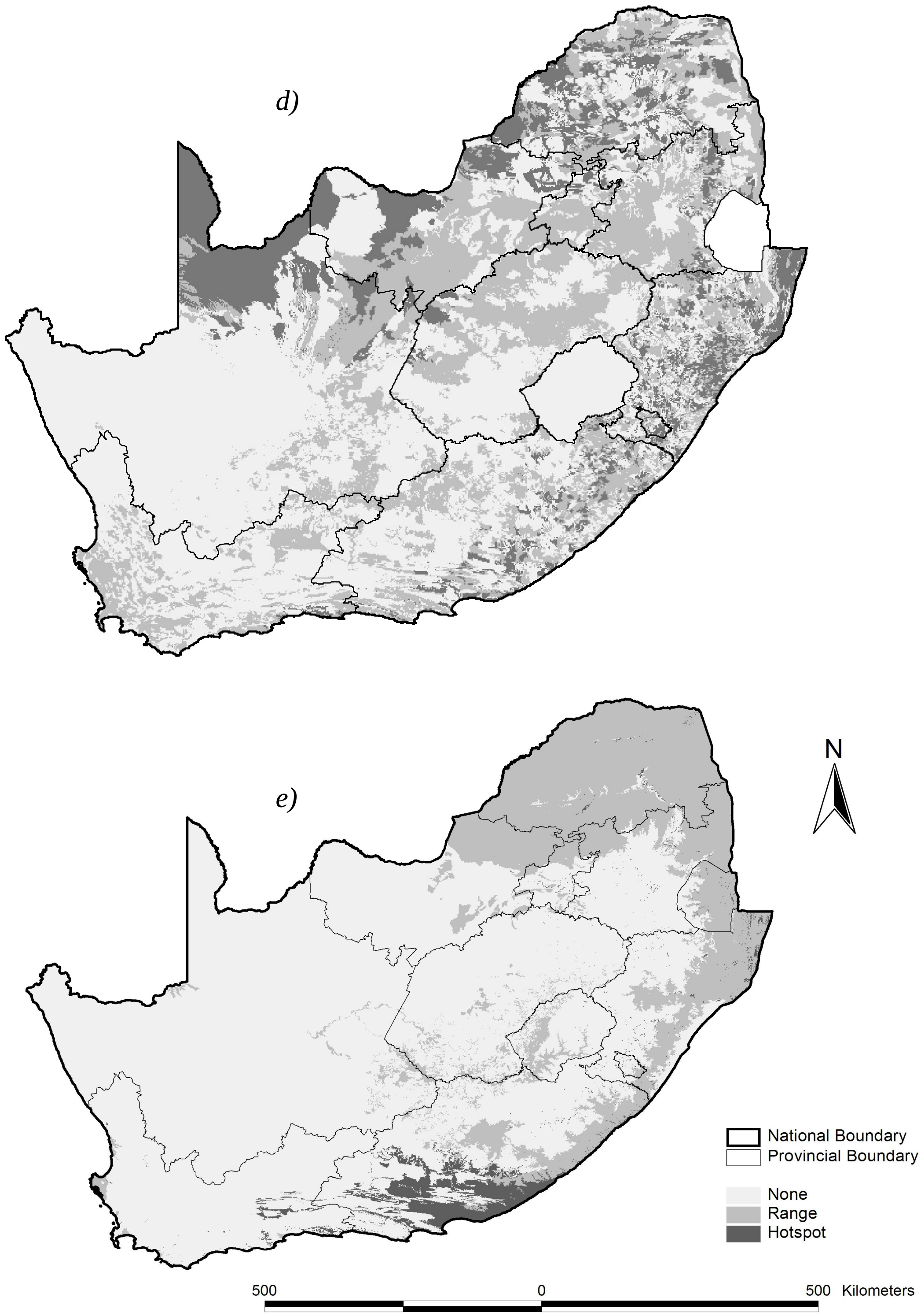

Figure 4.1: The extent of the range and hotspots of ecosystem services in South Africa used in this study. (a) Surface water supply (b) water flow regulation (c) soil retention (d) soil accumulation (e) carbon storage.

4.2 Method

4.2.1 Study Area

South Africa covers approximately 1.22 million km^2 of semi-arid landscape, with an annual mean rainfall that varies between 50 and 3000 mm (±450 mm). A significant proportion of the country's 47 million people (StatsSA, 2005) live in rural areas, with livelihoods that depend directly on ecosystem services. The country is exceptionally rich in biodiversity with high levels of endemism, containing three global biodiversity hotspots (Mittermeier et al., 2005).

4.2.2 Data

4.2.2.1. Biodiversity

Biodiversity data used in this study were compiled and used in the NSBA for South Africa (Reyers et al., 2007) and comprised information on species, vegetation, biomes and biodiversity priorities for the country. The species distribution data were at the resolution of a quarter-degree square (QDS; ≈700km^2). Species richness for various taxa were assembled for overlap and correlations analyses. I calculated species richness hotspots for birds, mammals, frogs, butterflies, endemic plants, threatened plants and all animals combined. The plant database extracted from the NSBA comprised two data sets: endemic plants and threatened plants (see Reyers et al., 2007). I delineated "richness hotspots" as the richest 10% of grid cells for each taxon (Orme et al., 2005).

Vegetation and biome data were obtained from the recent vegetation map of South Africa (Mucina and Rutherford, 2007). Vegetation diversity was calculated as the number of vegetation types found per quarter-degree square (QDS). The vegetation map recognises 441 vegetation types in total and nine biomes. Vegetation types per QDS ranged from 1-17. The 10% of grids with the most vegetation types per QDS were defined as vegetation-diversity hotspots.

The NSBA contained a map of biodiversity priorities. I assess the extent to which these priority areas overlap with the ecosystem service ranges and hotspots. The priority map integrated information on several levels of biodiversity (species, ecosystems, and processes), existing conservation efforts, gaps in target achievement, and pressures facing biodiversity

(Reyers et al., 2007). The NSBA included maps of carbon storage and water production as areas of importance for the maintenance of ecological processes in the priority selection.

4.2.2.2. Ecosystem services

Ecosystem services were selected on the basis of national importance, relevance to conservation planning, and availability of data. Thresholds were used to map the ranges and hotspots of each service. A brief description of the mapping of these services appears below (further details appear in chapter 3).

Surface water supply: In South Africa, water resources are mapped in water management areas termed catchments. Catchments, areas of land drained by a single river and its tributaries (DWAF, 2004) are classified into primary, secondary, tertiary and quaternary. Runoff was used as a surrogate for water supply (van Jaarsveld et al., 2005). Runoff is the total water yield from a catchment and includes surface and subsurface flow.

Water flow regulation: Water flow regulation is the storage component of water services and is highly dependent on ground water. Data on the percentage contribution of groundwater to base flows per quaternary catchment were extracted from DWAF (2005) and used to map water flow regulation.

Carbon storage: The carbon storage map used in this study was from the NSBA. Studies in South Africa have shown that some of the country's semi-arid ecosystems store exceptionally large amounts of carbon. Experts were able to use this and other information to define carbon storage potentials for different vegetation types.

Soil accumulation: This is a process of soil formation directly linked to the accumulation of soil organic matter. Soil accumulation was modelled as a function of soil depth and leaf litter accumulation index. Data on soil depth was obtained from Schoeman et al. (2002). I used Schulze's (2004) index of litter accumulation which estimates the soil surface covered by litter based on observations in a range of grasslands, woodlands and natural forests.

Soil retention: This is the ability of natural vegetation to curb erosion by holding onto soil. I modelled soil retention as a function of vegetation cover and erodibility. Schoeman et al.

(2002) modelled soil erodibility for the whole of South Africa based on soil structure, geology, water, wind and slope. Data on vegetation potential to curb erosion was based on expert knowledge.

4.2.3 Analysis

Three different methods (overlap, coincidence analysis and correlations) commonly used to assess spatial congruence were applied in this study (Prendergast et al., 1993; van Jaarsveld et al., 1998; Orme et al., 2005; Chan et al., 2006).

Overlap analysis: The percentage of each ecosystem service range and hotspot in each biome was measured. Both the biomes and the ecosystem services were converted to fine scale grid (0.01 degrees). The two grids were overlaid using geographic information system (GIS) and the amount of ecosystem service range or hotspot present in each biome was estimated and expressed as a percentage of total range or hotspot. The percentage of biome containing ecosystem services hotspot was also estimated. Proportional overlap (Prendergast et al., 1993), a measure of area shared between two entities expressed as a percentage of the one with the smallest area, was used to measure overlap between ecosystem service hotspots and species and vegetation diversity hotspots and biodiversity priorities. Hotspots of ecosystem services were assigned to QDSs for overlay with species data. However, because of the coarse scale of the species data and the potential for overestimating overlap, only QDSs where the hotspot of a service covered more than 10% of the QDS were included in this analysis. The number of QDSs containing both species and service hotspots was expressed as a percentage of the one with the smallest total number of QDS.

Comparison of species richness: Ecosystem services were summarised per QDS and those containing any amount of the hotspot of each ecosystem service were identified. The number of species per taxon and the number of all species were evaluated in each QDS identified as an ecosystem service hotspot. These numbers were used to determine whether ecosystem service hotspots contained more species than would be expected by chance. To generate an appropriate null expectation, a random sample (equal to the area of each ecosystem service hotspot) was repeatedly (10,000 times) drawn from 2014 QDS in the country, generating a distribution of random species. A p-value was obtained by comparing this distribution to the actual species captured by the ecosystem service hotspot. This analysis was repeated for vegetation diversity in which the mean vegetation diversity in the hotspot of each service was compared with that expected by chance.

Correlations: All ecosystem service data were summarised per QDS and the medians were obtained. Spearman correlations were run to determine relationships between species hotspots, vegetation diversity hotspot and ecosystem services median per QDS.

4.3 Results

It is clear that the larger biomes, grassland and savanna, contain significant percentages of almost all ecosystem services (Table 4.1). The fynbos and the Albany thicket biomes contain significant amounts of the water and carbon storage services. No ecosystem service range or hotspot was found in the desert biome. About 80% of the forest biome was important for the range and hotspot of carbon and at least 50% of this biome contributed to the range or hotspot of the water and soil services; but the forest biome is small compared to other biomes which resulted in small percentages of services in the biome.

Results from proportional overlap analysis showed that soil accumulation and soil retention had a relatively high overlap with all of the species hotspots (average of 43% and 36% respectively) (Table 4.2). The lowest overlap with these two services was with threatened plants. Water flow regulation and surface water supply both showed moderate overlap with species richness hotspots. Of all the ecosystem services, carbon storage exhibited the lowest overlap with species richness hotspots. Vegetation diversity hotspots showed a slightly higher overlap with carbon storage compared to other services. Overlap between biodiversity priorities and ecosystem services hotspots were generally high.

Results in Table 4.3 suggest that mean species richness is higher in hotspots of some ecosystem services than would be expected by chance. This was most evident in the hotspots of water flow regulation and soil accumulation. The number of bird and mammal species in the water flow regulation hotspot was almost twice as large as random expectation. On the other hand, the carbon storage hotspots did not include exceptional amounts of any of the taxa. However, carbon storage hotspots had a relatively strong correlation with vegetation diversity and threatened plants. Correlation results, though positively significant, were generally weak (Table 4.4). The water services hotspots showed some consistency by showing significant but weak positive correlation with species hotspots for all taxa. Soil accumulation showed a stronger correlation with most species hotspots than other services especially with mammal richness.

Surface water supply and water flow regulation hotspots showed different levels of overlap with species richness hotspots and mean species richness between the two was also different compared to that expected by chance. Water flow regulation exhibited a higher level of congruence with all taxa hotspots (but see endemic plants and mammals) than surface water supply except for frog hotspot. This trend was also observed with the soil services. Soil accumulation showed a higher level of congruence with all taxa than soil retention for overlap, comparison of species richness and correlations analysis.

4.4 Discussion

Results from this study indicate that certain biodiversity facets co-occur with ecosystem services in South Africa, suggesting that opportunities exist for using ecosystem services as an additional rationale for biodiversity conservation. These opportunities exist mostly in the grasslands and savanna biomes where all five services could be used to justify conservation. Although these biomes are large, the relatively high percentage of these biomes contributing to ranges and hotspots of services indicates their importance in service delivery. Furthermore, the Nama Karoo (third largest biome in the country) did not contain a high percentage of service ranges or hotspots. Water regulation and supply may be used to justify conservation in the fynbos biomes and in the Indian Ocean coastal belt; carbon storage, in the Albany thicket biome; and the soil services, in Albany thicket and fynbos biomes. The forest biome is associated with many ecosystem services, in particular carbon storage (Myers, 1988; Turner et al., 2007), but this biome covers a very small area in South Africa and only a small percentage of the forest contained each ecosystem service. This does not imply that the forest is not important for ecosystem services. In the semi-arid biomes of the Nama and Succulent Karoo, these five ecosystem services do not provide an additional rationale for biodiversity conservation.

Table 4.1: Percentage of each ecosystem service range and hotspot in each biome in South Africa. Bold entries indicate the top three biomes with the greatest percentage for each service. The percentage of the biome containing hotspot of each service is shown in brackets.

	Surface water supply		**Water flow regulation**		**Soil retention**		**Soil accumulation**		**Carbon storage**	
	Range	**Hotspots**	**Range**	**Hotspots**	**Range**	**Hotspots**	**Range**	**Hotspots**	**Range**	**Hotspots**
Albany Thicket	0.8	0.0	1.6	0.8 (2.9)	4.0	**6.4** (25.3)	2.4	3 (16.4)	**9.2**	**77.9** (89.6)
Desert	0	0	0	0	0	0	0	0	0	0
Forest	0.2	0.2 (11)	0.2	0.2 (23)	0.2	0.3 (29.2)	0.1	0.3 (38.4)	0.3	2.3 (79.8)
Fynbos	**7.9**	6.7 (5)	**13.1**	**13.1** (17)	**12.1**	0.62 (0.9)	6.4	0.8 (1.5)	1.5	4.0 (1.7)
Grassland	**63**	**66.8** (12)	**48.33**	**46.9** (15)	**31.8**	**25.6** (9.2)	**31.7**	**12.0** (5.7)	**11.6**	**5.7** (0.6)
Indian Ocean Coastal Belt	5.4	**10.6** (41)	2.9	4.3 (30)	1.9	3.3 (25.6)	2.3	**4.6** (47)	3.1	**4.0** (9.2)
Nama- Karoo	0.1	0.0	0.3	0.2 (0.1)	10.8	0.7 (0.4)	**10.4**	1.0 (0.6)	1.2	1.2 (0.2)
Savanna	**22.5**	**15.8** (2)	**32.7**	**33.1** (9.5)	**37.3**	**63.0** (20)	**46.1**	**78.3** (32.8)	**73.0**	4.0 (0.4)
Succulent Karoo	0.1	0	0.8	1.4 (1.8)	1.9	0.1 (0.1)	0.5	0	0.1	0.9 (0.4)
Total	100	100	100	100	100	100	100	100	100	100

Table 4.2: Proportional overlap between species richness hotspots, vegetation diversity hotspot, biodiversity priority areas and ecosystem service hotspots.

	Water flow regulation	**Surface water supply**	**Soil retention**	**Soil accumulation**	**Carbon storage**
Birds	37.1	19.5	34.2	46.5	13.4
Frogs	24.6	30.6	47	57.9	11.8
Butterfly	34.7	24.2	43.2	47.7	21.0
Mammals	31.7	25.8	47.7	63.8	8.4
Animals combined	32.7	25.3	45.7	58.3	9.2
Threatened and endemic plants	34.0	27.4	36.5	39.5	22.7
Threatened Plants	24.7	18.4	18.2	17.2	17.6
Endemic plants	29.5	17.9	22	24.0	37.0
Vegetation diversity	34.0	23.2	26.7	27.7	41.2
Biodiversity priority	70.7	70.0	55.7	42.0	65.5

Table 4.3: Ratio of mean species richness and vegetation diversity in ecosystem services hotspots compared to the mean of randomly drawn samples. Entries in brackets are not significant at p<0.05.

	Water flow regulation	**Surface water supply**	**Soil retention**	**Soil accumulation**	**Carbon storage**
Birds	1.803	(1.004)	1.026	1.19	(1.002)
Frogs	1.505	1.01	1.025	1.061	(1.002)
Mammals	1.714	(1.012)	1.025;	1.155;	(1.002)
Butterflies	1.577	1.011	1.025	1.151	(1.002)
Animals combined	1.26	1.008	1.006	1.06	(1.002)
Threatened and endemic plants	1.46	1.01	(1.007)	1.07	(1.002)
Endemic plants	1.28	(1.004)	(1.001)	(0.977)	(1.002)
Threatened plants	1.37	(0.997)	1.020	1.115	(1.002)
Vegetation diversity	1.73	(0.998)	(1.002)	1.21	(1.002)

Table 4.4: Spearman rank correlation between species richness and ecosystem service hotspots. Only significant r values are reported. Correlations are significant at $p<0.05$

	Water flow regulation	Surface water supply	Soil retention	Soil accumulation	Carbon storage
Birds	0.22	0.1	0.1	0.14	
Frogs	0.1	0.23	0.19	0.23	
Butterfly	0.19	0.15	0.16	0.15	0.1
Mammals	0.17	0.18	0.19	0.27	
Animals combined	0.18	0.16	0.18	0.24	
Threatened and endemic plants	0.18	0.18	0.11		0.11
Threatened plants	0.1	0.1		-0.08	0.06
Endemic plants	0.14	0.18			0.23
Vegetation diversity	0.18	0.13			0.26

Our finding of high overlap between some ecosystem services and biodiversity priorities supports findings by Turner et al. (2007) and Chan et al. (2006). The reason for the high overlap in our case is unclear. Although total area may be a contributing factor (NSBA priorities for South Africa cover almost 50% of the land), the overlap was also higher than expected by chance for hotspots (Table 4.3). The high species richness found in hotspots of many ecosystem services supports the highly debated positive relationship between species richness and ecosystem functions (Mittelbach et al., 2001). In chapter 3, I found a strong positive correlation between the ecosystem services included in this study and primary productivity suggesting high productivity areas are high in both species and ecosystem services especially at the scale of this study. Several studies have shown a positive relationship between species richness and productivity (Balmford et al., 2001; van Rensburg et al., 2002; Chown et al., 2003). These results were also supported by the moderate to high overlap and the positive correlations between some ecosystem services and species hotspots. However, our finding of moderate to high overlap of biodiversity hotspots and ecosystem services could have been overestimated due to our mapping of species hotspots. I used the top 10% grid cells compared to the more traditional five percent or less in other studies (van Jaarsveld et al., 1998; Orme, et al., 2005).

The difference in the level of congruence between various water and soil services with species and vegetation diversity hotspots and the low concordance between these services is an indication that these services are driven by different variables and should each be considered separately in conservation assessments. Water flow regulation showed a stronger concordance with species richness and vegetation diversity hotspots compared to surface water supply, however surface water supply is the only water service considered in conservation assessments in South Africa (Reyers et al., 2007). The soil services were also not consistent in their relationship with species richness and vegetation diversity. These results suggest that no single biodiversity measure (e.g. vegetation types or species richness) can be used as a surrogate for ecosystem services and vice versa. This study seems to indicate that as many surrogates of ecosystem services as possible should be considered due to the low level of concordance between ecosystem services. This is similar to the issues debated over a decade ago in biodiversity surrogacy (see Lombard et al., 2003 for an example) and it appears that no one rule applies to ecosystem services and their relationships with biodiversity. Overlap between biodiversity pattern and ecosystem services may vary from service to service depending on the biodiversity data used and the scale of study. These results can not be conclusive until congruence is examined within an exhaustive list of ecosystem services and biodiversity at different resolutions. They should be interpreted with caution because not only is the overlap variable, but biodiversity and ecosystem services have different values and need different management approaches.

Including ecosystem services in conservation assessments and finding areas of synergy may have some benefits, but the continuous provision of services will require appropriate targets for ecosystem services that will ensure the continuity of the functions (Kremen and Ostfeld, 2005). Such targets may not be achieved in conservation assessments geared towards biodiversity alone (Chan et al., 2006) and may require a separate plan for ecosystem services. In addition, such a plan must consider threats facing each service as is the case for biodiversity where critically endangered features are given priority (Sisk et al., 1994; Balvanera et al., 2001). Successful management of ecosystem services and biodiversity however demands a multidisciplinary approach which takes many factors into consideration, and involves all stakeholders (Cowling et al., 2008). At present, planning and management of these resources is carried out by different organisations in South Africa with water resources managed separately from soils and biodiversity. An integrated approach is necessary so that the management of one does not deplete the other. However, the need for integration in order to improve implementation is starting to emerge. An example is the Working for Water program whereby the department of water affairs and forestry (DWAF) is joining forces with conservation agencies to manage invasive alien plants both for biodiversity and water supply while creating employment (Turpie et al., 2008). This type of approach provides a win-win situation and reduces the cost of implementation.

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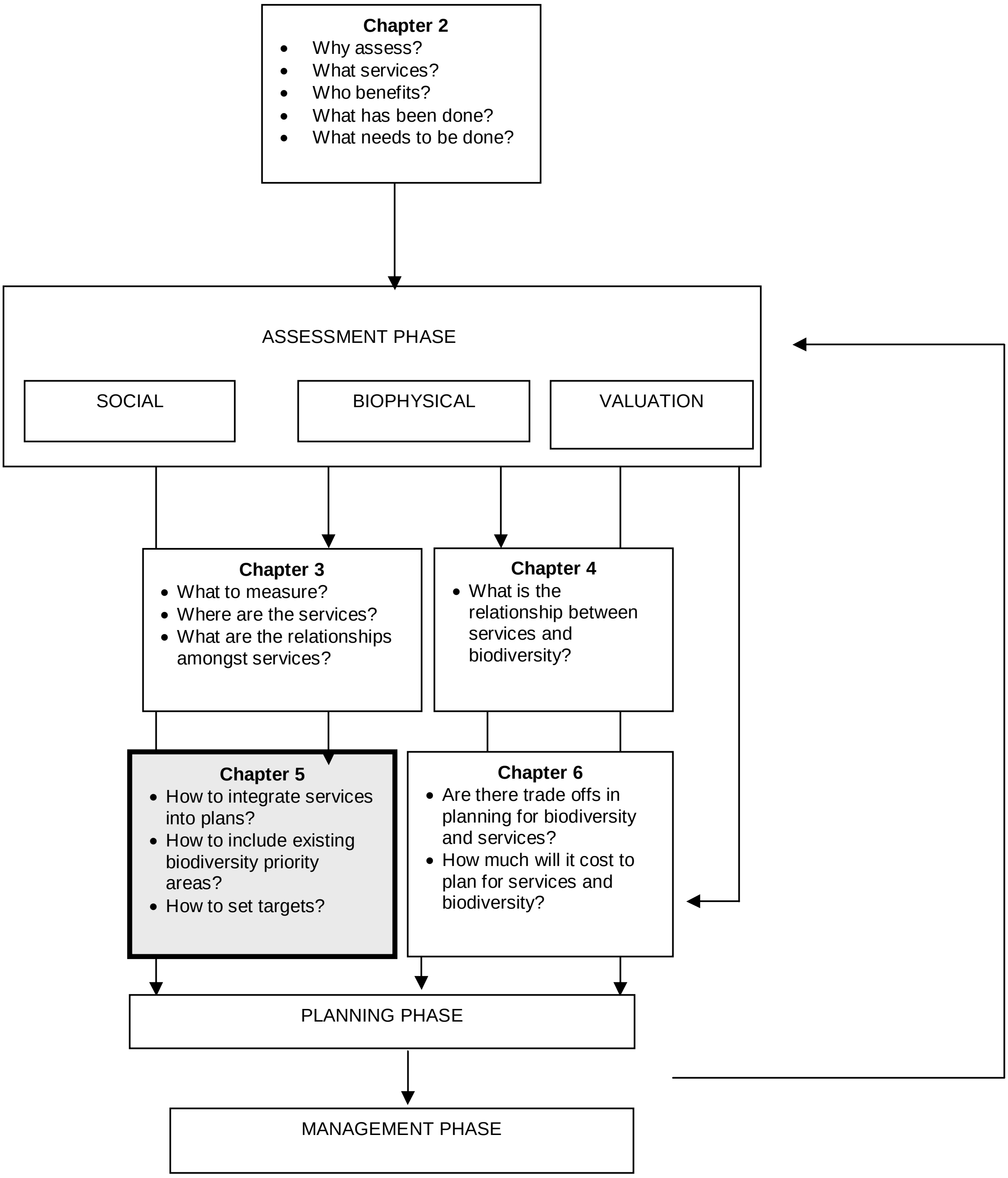

Chapter 2
Why assess?
What services?
Who benefits?
What has been done?
What needs to be done?
ASSESSMENT PHASE
SOCIAL
BIOPHYSICAL
VALUATION
Chapter 3
What to measure?
Where are the services?
What are the relationships amongst services?
Chapter 4
What is the relationship between services and biodiversity?
Chapter 5
How to integrate services into plans?
How to include existing biodiversity priority areas?
How to set targets?
Chapter 6
Are there trade offs in planning for biodiversity and services?
How much will it cost to plan for services and biodiversity?
PLANNING PHASE
MANAGEMENT PHASE

CHAPTER 5. IDENTIFYING PRIORITY AREAS FOR ECOSYSTEM SERVICE MANAGEMENT IN THE GRASSLANDS OF SOUTH AFRICA

ABSTRACT

Grasslands are the source of many ecosystem services required to support human well being. In addition to the many services provided by grasslands, they are home to diverse fauna and flora. Conversion of grassland to other forms of land use threatens this biodiversity and many ecosystem services. In response studies around the world have identified geographic priority areas to direct conservation efforts for biodiversity. However, no such studies exist for the identification of priority areas for ecosystem services. A critical barrier to ecosystem service planning and prioritisation is the lack of spatial data for services, as well as the setting of targets for ecosystem services. This study maps five ecosystem services (carbon storage, surface water supply, water flow regulation, soil accumulation and retention) and identifies priority areas where management efforts should be focused for individual ecosystem services, as well as a suite of services at a range of target levels. It also aligns ecosystem service priority areas with biodiversity priorities and assesses the extent to which such priorities have been compromised by different land uses. Planning for individual services showed that few catchments were irreplaceable. The number of catchments needed to conserve ecosystem service increased with target levels. Catchments in the province of KwaZulu Natal were important in all scenarios. Irreplaceability values were low when planning for ecosystem services only, suggesting flexibility in the choice of catchments to be included in the priority maps, however when attempting to align these priorities with biodiversity priority areas this flexibility decreased. Transformation and degradation levels of ecosystem services priorities were similar to those for the entire biome (30%). The implications of these results for managing ecosystem services in the grasslands are discussed.

Keywords: Conservation planning, grasslands, ecosystem services, priority areas, integration

5.1 Introduction

Ecosystems provide many benefits to humans. Grasslands, originally covering approximately 25% of the land surface on Earth, are no exception (Graetz, 1994). Grasslands are the source of many ecosystem services required to support human well being. They sequester carbon as soil organic matter stored mostly below ground (Burke et al., 1989, Sala and Paruelo, 1997). Many communities use grasslands as grazing land or for the collection of medicinal plants, fruits, thatch grass, or hunting to name but a few (Sala and Paruelo, 1997; Friday et al., 1999; Dzerefos and Witkowski, 2004). In South Africa approximately 30% of all plants sold in traditional medicine markets grow in grasslands (Williams et al., 2000). An important ecosystem service in the grassland is that of water supply. Most of South Africa's water originates from mountain ranges in the grassland. The biome plays a crucial role in the hydrological cycle, by reducing immediate runoff and thus erosion; and by storing runoff as groundwater or in wetlands, which is slowly released during the year creating a steady water supply (Kotze and Morris, 2001).

In addition to the many services provided by grasslands, they are home to a diverse fauna and flora. Globally, grasslands house many important species including 15% of the world's Centres of Plant Endemism, 11% of Endemic Bird Areas and 29% of ecoregions with outstanding biological distinctiveness (White et al., 2000). South African grasslands host a very high diversity of plant species, second only to the Cape Floral Kingdom (greater at a $1000m^2$ resolution; O' Connor and Bredenkamp, 1997). A high degree of endemism also occurs with nearly half of South Africa's 34 endemic mammals found in the Grassland biome. The biome houses 22% of South Africa's endemic reptiles, a third of threatened butterflies and 5 of the 17 Ramsar wetlands in South Africa (http://www.sawac.co.za/articles/GrasslandFacts.htm). The grasslands are also an area of importance to freshwater biodiversity. Thirty-eight river ecosystems have been identified within the grasslands in South Africa which are critical to conserving fresh water and its biodiversity (Nel et al., 2007).

Declines in grassland area, through conversion to other land uses (croplands) or through poor land use management practices (overgrazing), threaten both biodiversity and many ecosystem services. Grasslands across the world are one of the biomes most impacted on by humans and their activities. The Global 200 ecoregions assessment (Olson and Dinerstein, 1998), as well as the report drawn up by the World Resources Institute in their Pilot Assessment of Global Ecosystems (White et al., 2000) reported declines in grassland condition, biodiversity and ecosystem service delivery and highlighted grassland conservation as a major concern. The recently completed Millennium Ecosystem Assessment highlighted that while most global biomes had lost 20 – 50% of their area to cropland conversion, temperate grasslands lost more than 70% of their natural cover by 1950 and a further 15.4% since then (MA, 2005). In South Africa about 30% of the grassland biome is transformed (Fairbanks et al., 2000). These results make grasslands one of the greatest conservation priorities for biodiversity and ecosystem services at all scales.

The need to safeguard ecosystem services was emphasized by reports of widespread decline in ecosystems and the degradation and unsustainable use of ecosystem services (WRI, 2001; MA, 2005). Safeguarding ecosystem services requires the mainstreaming of ecosystem services for implementation and management (see Cowling et al., 2008). An important step toward achieving this goal is a biophysical assessment of ecosystem services that quantifies and maps ecosystem services, identifies priority areas where conservation efforts should be focused and assess threats facing such areas. Areas that are important for providing services must be identified and managed to ensure sustainable delivery of services (Balvanera et al., 2001; MA, 2005). Ecosystem services are often used to justify biodiversity conservation and can facilitate implementation. There are calls to include ecosystem services into biodiversity conservation plans (Singh, 2002; see chapter 2 for review). This approach seems promising, but is challenging. As highlighted in chapter 2 of this thesis, ecosystem service planning can benefit from the two decades of research and development that has gone into the field of conservation planning. The data, methods, tools and lessons learnt in this field provide a valuable starting point for ecosystem service planning. However, integrating multiple features (e.g. species, habitat, ecological processes and services) into one plan is challenging and as yet unprecedented. Furthermore, ecosystem services are not the same as biodiversity and may need different management approaches from those for biodiversity.

Many studies around the world have identified priority areas where conservation efforts for biodiversity should be focused. A few of these studies have integrated ecosystem services into such biodiversity assessments to some extent (see chapter 2 for review). More recently, studies that have mapped ecosystem services have focused mainly on assessing congruence with biodiversity (Chan et al., 2006; Tuner et al., 2007; Naidoo et al., 2008) without necessarily focusing on identifying priority areas for the management of ecosystem services and biodiversity.

This study addresses this gap by:

1. Identifying priority areas for managing ecosystem services in the grasslands
2. Aligning the goal of biodiversity conservation and ecosystem service management by integrating biodiversity priorities into ecosystem service planning.
3. Evaluating the extent to which these priorities have been compromised by land cover change.

5.2 Methods

5.2.1 Study area

The South African grassland biome covers an area of about 339 240 km^2 (373 990 km^2 including Lesotho and Swaziland) and contains 80 vegetation types (Mucina and Rutherford, 2007). The biome overlaps with several of South Africa's provinces and magisterial districts (Figure 1.2). Provinces with the greatest cover of grassland vegetation in South Africa are the Free State (86%), Gauteng (67%), Mpumalanga (64%) and KwaZulu Natal (59%).

The rainfall gradient in the grasslands ranges from 400 to >1200 mm yr^{-1} altitude ranges from sea level to >3300 m, and soil types range from humic clays to poorly structured sands (O'Connor and Bredenkamp, 1997). The grasslands are the most productive biome in terms of agriculture in South Africa (Mentis and Huntley, 1982). In an assessment based on Low and Rebelo's (1996) definition of the Grassland biome (including Lesotho and Swaziland an area of 334001 km^2) and the 1996 land cover data, Fairbanks et al. (2000) illustrate that 29.2% of the grasslands has been converted to some other form of land use (cultivation – 23.48%; forest plantations – 3.35%; Mines and quarries 0.32%; Urban – 1.92%; improved grassland – 0.13). Agriculture is mostly irrigated, making water one of the most limited resources in South Africa. The regulation of the quality and quantity of water are crucial ecosystem services in the grasslands with far reaching effects into the rest of South Africa (Kotze and Morris, 2001).

5.2.2 Data

Three groups of data were used in this study: five ecosystem services, freshwater and terrestrial biodiversity priorities. Ecosystem service layers were mostly extracted from those derived in chapter 3 of this study. Freshwater biodiversity data were extracted from the rivers component of the National Spatial Biodiversity Assessment for South Africa (NSBA) (Nel et al., 2007). Data on terrestrial biodiversity were extracted and refined from NSBA (Reyers et al., 2007). These databases and their analysis are described in the subsequent sections.

5.2.2.1. Ecosystem services

Five ecosystem services were considered in this study and include: surface water supply, water flow regulation, carbon storage, soil retention and accumulation. These services were selected based on their importance in the grassland. Data on water supply and flow regulation, soil retention and accumulation were extracted from those derived in chapter 3 of this thesis. Carbon storage data used in this study was extracted from a global data on carbon and was different from those derived in chapter 3. A few studies in South Africa and elsewhere have measured ecosystem services provision together with beneficiaries (van Jaarsveld et al., 2005; Naidoo and Ricketts 2006). However, maps used in this study, only consider the biophysical potential for provision of ecosystem services because the services considered in this study are almost needed everywhere as soil and water resources are highly stressed in South Africa and carbon sequestration benefits the global community. Furthermore, maps of beneficiaries are not readily available at the national scale, as this is a complex phenomenon with beneficiaries deferring both in temporal and spatial scales. Below is a brief description of how the services were mapped.

Surface water supply: Surface water supply is a function of the quantity and quality of water available for direct use to humans. In South Africa, water quality and quantity are correlated. High runoff areas tend to have good water quality and so in chapter 3 of this thesis, I used runoff as a surrogate to map water supply which was extracted for this study. Management of these areas will maintain or improve water quality because when they are kept in a good condition they yield high quality water, with the lowest possible soil erosion, nutrient and sediment loss (Scanlon et al., 2007).

Water flow regulation: Water flow regulation is the storage component of the water services and is a function of the contribution of groundwater to base flow. Degradation or transformation of such areas could deplete groundwater reserves or lead to salinization of such reserves. In chapter 3, I mapped water flow regulation as the percentage contribution of groundwater to base flow per quaternary catchment from DWAF (2005) at a national scale. Water flow regulation for the grasslands was extracted from the national data.

Carbon storage: The retention of carbon stored above or below the ground has the potential to mitigate climate change impacts. Data on carbon storage (kg/m^2) were extracted from the ISRIC-WISE Global Data Set of Derived Soil Properties, Southern Africa (GSDT, 2002). This dataset contained data on soil-carbon density, total nitrogen density, profile of available water capacity amongst others. Carbon data for South Africa were clipped from the southern Africa dataset using Geographic Information Systems (GIS). These data were converted into tonnes of carbon stored per hectare.

Soil retention: Areas where vegetation cover retains soils need to be managed carefully to ensure the continuous delivery of the services of land productivity and prevent damage from erosion through sedimentation and eutrophication of nearby rivers. Soil retention was mapped as a function of vegetation cover and soil erosion potential (see chapter 3). Data on soil erodibility were obtained from Schoeman et al. (2002). They modelled soil erodibility for the whole of South Africa based on soil structure, geology, water, wind and slope. Data on vegetation potential to curb erosion was based on expert knowledge. In this study, soil retention for the grassland biome was clipped from maps generated in chapter 3 of this thesis.

Soil accumulation: In chapter 3 of this thesis, I mapped soil accumulation as a function of soil depth and leaf litter accumulation index. The accumulation of soil organic matter is an important process of soil formation which can be badly affected by habitat degradation and transformation (de Groot et al., 2002). Data on soil depth was obtained from Schoeman et al. (2002). Data on littler was extracted from Schulze’s (2004) index of litter accumulation which estimates the soil surface covered by litter based on observations in a range of grasslands, woodlands and natural forests. Data on soil accumulation was extracted from chapter 3 for the grassland biome only.

5.2.2.2. *Biodiversity data*

Terrestrial biodiversity

Three sets of biodiversity data were used in this study: species, habitat and biodiversity priority maps. The species and habitat priority maps were extracted from the national spatial biodiversity assessments (NSBA) for South Africa. The NSBA scored each quarter-degree square (QDS - NSBA planning unit) from 0-100 based on endemic and threatened species irreplaceability. Data on habitat consisted of scores (0-100) for each planning unit (QDS) based on the vegetation irreplaceability, threatened status and protection level (more detail can be found in Reyers et al., 2007). These two layers of species and habitat priority scores were used as cost layers while planning for ecosystem services (see below). The overall biodiversity priority map for the grassland biome was only used to compare overlap with outputs from this study. This map is based on the species and habitat priority maps described above, as well as maps of the spatial components of ecological and evolutionary processes (extracted from Reyers et al., 2007). These priority maps were combined and refined in this study by removing areas converted to other land uses based on data from the National Land Cover 2000 and road data (Reyers et al., 2001) to map the distribution of overall biodiversity priority areas. Although the NSBA was conducted at a national scale, using national-scale data, the size of the grassland biome and the absence of finer scale data for the biome as a whole implied that the NSBA data would be appropriate for the grasslands.

Freshwater biodiversity priority areas

Data on freshwater biodiversity were extracted from the NSBA (Nel et al., 2007) and refined for the purpose of this study. The freshwater component of the NSBA did not get as far as identifying priority areas for freshwater biodiversity; instead it focused on identifying threatened rivers and rivers that were not adequately protected. In identifying threatened rivers, it used desktop estimates of present ecological status from the national Water Situation Assessment Model (WSAM; Kleynhans, 2000) to classify main river integrity (Nel et al., 2007). Main rivers were defined as the 1:500 000 rivers which pass through a quaternary catchment into a neighbouring quaternary catchment (Midgely et al., 1994). These main rivers were grouped into river types based on geomorphology and hydrological descriptors. Critically endangered river types (those with less than 20% of the main river length intact) from the NSBA were the focus of this study. The NSBA revealed that tributaries were generally in a better condition than the main rivers. For the purposes of this study, intact tributaries (> 75 % length intact) of the critically endangered river types longer than 10km were selected as priority areas for conservation efforts. Catchments that contained any of these selected tributaries were identified as fresh water conservation priorities and were assigned a score of 100.

5.2.3 Analysis

Planning unit and data preparation

Catchments were chosen as planning units (the building block of the planning domain) because in South Africa water resources are managed in catchments (watersheds). A catchment is defined as the area of land that is drained by a single river system, including its tributaries (DWAF, 2004). The water services were already mapped at the catchment resolution. The water services were summarised in litres per catchment. The remaining three ecosystem services data were overlaid with the catchments layer. In chapter 3 of this thesis, all services were mapped as ranges and hotspots. The range of an ecosystem service was defined as the areas of meaningful supply while hotspots were defined as areas which provide large proportion of a particular service. Only the hotspot of soil retention and accumulation were considered in this study and the total area per catchment of each of the hotspots was calculated. Total amount of carbon per catchment was calculated in tonnes. At present, the effect of transformation on ecosystem services is not well known and it is difficult to estimate amount of service produced while taking transformation into consideration. I therefore used the potential ecosystem service per catchment rather than the actual.

Identifying ecosystem service priorities within the grassland biome

Planning for ecosystem services is still in its infancy and no spatially-explicit decision support systems have been developed to aid in planning. The software platforms MARXAN (Possingham et al., 2000) and C-plan (Pressey, 1999) are commonly used by conservation biologists to plan for biodiversity. These platforms are also suitable for application in planning for managing ecosystem services. I used simulated annealing within MARXAN version 1.8.2, which selects sets of priority areas that meet targets for features (usually biodiversity features like species) at a minimal cost (Possingham et al., 2000). The number of runs and iterations used for all analysis was 1000 and 1000 000 respectively. Although 100 runs may be sufficient to produce good results, I chose 1000 runs to increase the distribution of irreplaceability values.

One of the characteristics of systematic conservation planning is its use of explicit, quantitative targets for biodiversity features (Margules and Pressey, 2000). Determining these targets for biodiversity features continues to be a challenge, and is a significant obstacle in adapting these algorithms for ecosystem service planning. Biodiversity targets can be based on ecological data (e.g. species-area curves, or population viability analyses) while some are social or political (e.g. IUCN's 10% recommendation for protected area extent). Similarly, targets for ecosystem services could be based on biophysical thresholds or societal need or goals. For example, the natural area needed for pollination; the area of a watershed needed for water purification and the amount of clean water needed by a village per year. Van Jaarsveld et al. (2005) used current demand as a way of setting ecosystem service targets, however this approach does not consider future demand, or ecological thresholds and needs for these services (e.g. the "ecological reserve" in a river). Generally, this level of information is lacking and poses major challenges for planning for ecosystem services.

At present, there is a lack of information to inform the setting of targets for ecosystem services. In this study, I therefore explored a range of target levels when planning for ecosystem services separately and collectively. When planning for individual ecosystem services, targets were set as follows: 40% of total surface water supply and water flow regulation (following Chan et al., 2006), and 40% for carbon storage. A slightly higher (50%) target was set for soil accumulation and soil retention because only hotspots were considered and the nature of the services demands that area within the hotspots should be kept in a natural condition. Catchment area was used as a cost layer to bias selection to favour smaller catchments.

I first analysed patterns of irreplaceability for individual services - these are maps of the importance of an area for a particular ecosystem service target. Areas of high irreplaceability are always needed to achieve the target, while areas of lower value can be swapped with one another to achieve the target. Although individual services can be planned for and managed separately, considering a range of services in one plan ensures efficiency in selecting areas that can be managed for multiple service delivery. The second set of analyses therefore combined all five services into one plan to explore the area needing management to meet different target levels across a suite of services. Below, I describe these target levels. In the third analysis, I integrated biodiversity information into one of the plans that considered all five services. A set of scenarios reflect different target levels and the integration of freshwater and terrestrial biodiversity into a plan for ecosystem services is described below.

Scenario 1: A 20% target of total supply was used for surface water supply, water flow regulation, and carbon storage with a 50% target for soil retention and accumulation hotspots. Catchment area was used as a cost layer to bias selection to favour smaller catchments.

Scenario 2: A 40 % target was set for surface water supply, water flow regulation and carbon storage. A 50% target was set for soil retention and accumulation hotspots. Catchment area was used as cost as in Scenario 1.

Scenario 3: A 60% target was set for surface water supply, water flow regulation and carbon. I targeted 50% of the total area of hotspots for soil retention and accumulation. Catchment area was used as cost as in Scenario 1.

Aligning ecosystem service priorities with biodiversity priorities

The integrated plan (Scenario 4 below) included freshwater and terrestrial biodiversity into ecosystem service planning. There are many ways to integrate biodiversity data into an ecosystem service plan. One could consider biodiversity surrogates such as species and habitat types or make use of already identified priority maps. At the inception of this study, there were many fine-scale plans in the region and the grassland biodiversity assessments mentioned above. These projects have already started channelling funding and resources to these identified priorities and thus I chose to align the ecosystem service priorities with these biodiversity priorities instead of using individual species or vegetation types. I used freshwater and terrestrial priorities while planning for ecosystem services such that where there were options between catchments, MARXAN would select a catchment that was already identified as a priority for biodiversity. I only considered species, habitat and threatened river types as priorities, because the overall terrestrial priorities contained ecological processes, some of which would overlap with the services considered in this study.

Scenario 4 (Integrated plan): The aim was to meet targets for ecosystem services in areas that are also priorities for conserving biodiversity, where possible. The targets in scenario 2 were maintained and terrestrial (species and habitat) and freshwater priorities were added to the cost layer. To ensure that planning units with higher biodiversity values were selected, I summed the priority scores for species (0-100), habitat (0-100) and freshwater biodiversity (100) and then calculated their inverse as a measure of “cost”. Thus, catchments which were priorities for many features would get a very low cost and were thus more likely to be selected.

All scenarios were run in MARXAN and results were summarised using two MARXAN outputs. First, the frequency with which catchments were selected was assessed from the 1000 runs per scenario. This frequency of selection serves as an estimate of irreplaceability defined as the likelihood that a given site will need to be protected to achieve a specified set of conservation targets (Pressey et al., 1994; Ferrier et al., 2000). I estimated the percentage of catchments selected for four irreplaceabilty classes (0-0-25, >0.25-0.5, >0.5-0.75, >0.75) for each scenario. Second, the best reserve system (best solution) was identified. This solution is the set of catchments selected to meet specified targets at a minimum cost. Proportional overlap was used to calculate overlap between ecosystems service priorities (best solution) from Scenarios 1-4 and freshwater and terrestrial biodiversity priorities (from the NSBA).

In the last set of analyses, the extent to which priorities from different scenarios and the integrated plan has been affected by land use was evaluated using three land cover classes (natural, degraded and transformed) from the national land cover information (Fairbanks et al., 2000). The total area under natural, degraded and transformed conditions was estimated for each catchment. This was summed for all catchments selected as part of the best solution for each scenario.

5.3 Results

In the first set of analyses, planning for individual ecosystem services showed that for carbon storage 30% of the catchments had irreplaceability values > 0.5. These areas were mostly in the forest vegetation in the eastern parts of KwaZulu Natal (KZN), the northern Free State (FS), the wetlands in southern Mpumalanga (MP) and southern Limpopo (LP) province. Fewer catchments in the study area had irreplaceability values > 0.5 for the water services (19% for surface water supply and 7% for water flow regulation). Irreplaceability maps of soil accumulation and soil retention showed that 15% and 19% of catchments had irreplaceability values > 0.5 respectively. Catchments in the eastern parts of KZN were mostly highly irreplaceable for water services, whereas catchments in the northern parts of the Eastern Cape and a few in Mpumalanga also had high irreplaceability for surface water supply. Relatively highly irreplaceable catchments for soil services were scattered all over KZN for both services and a few in LP and NW for soil accumulation. Maps of the best solution when planning for individual ecosystem services showed that 20% and 8% of all catchments are needed to conserve 40% of the total amount of water available for surface water supply and water flow regulation respectively. Most of these catchments are in the eastern and northern parts of the grassland biome (Figure 5.1). About 43% of all catchments were needed to conserve 40% of the carbon stored in the region. To conserve 50% of the soil services, 18% and 15% of catchments were selected for soil retention and accumulation respectively.

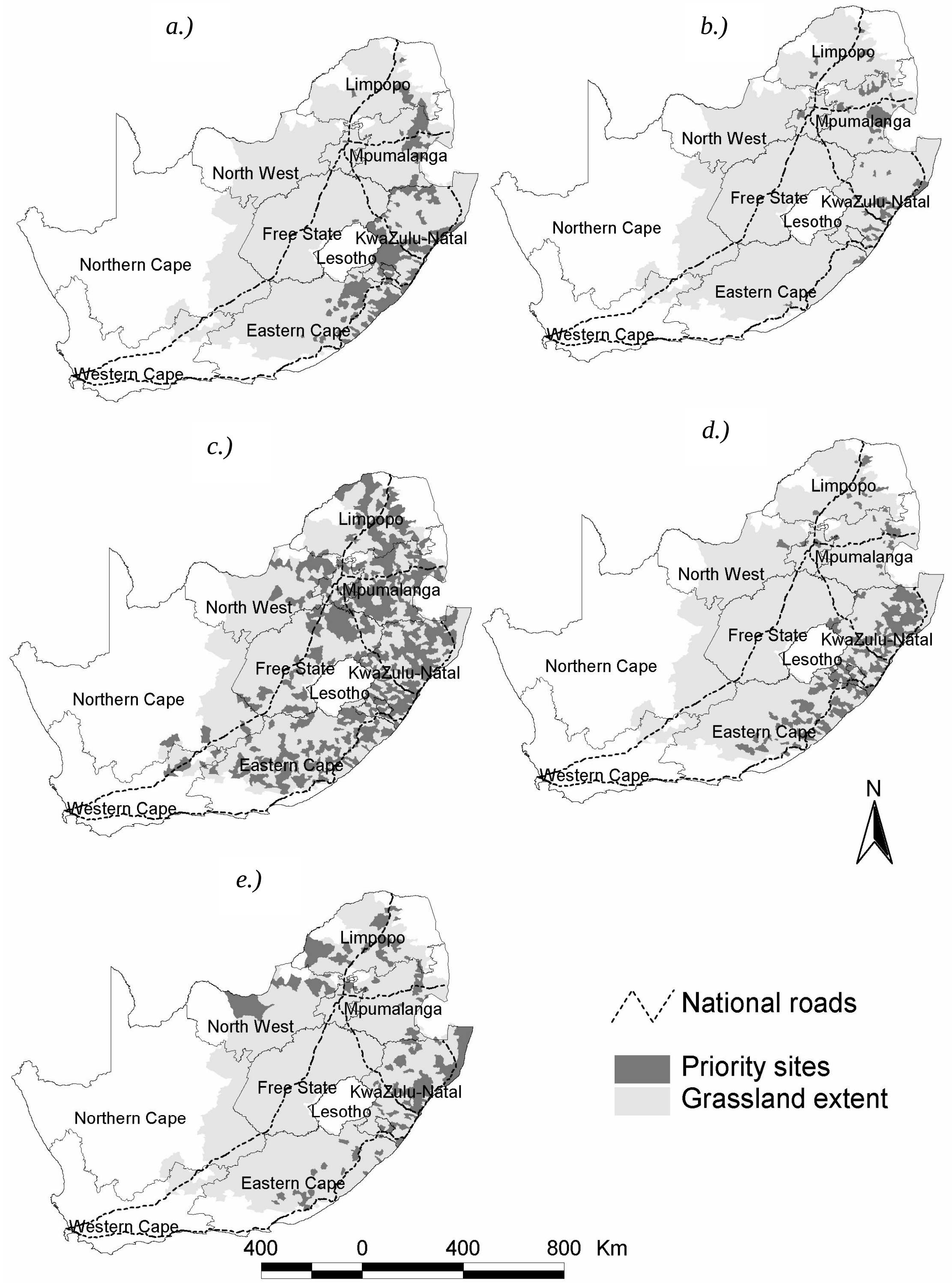


Figure 5.1: Priority areas of each service planned for separately (a) surface water supply (b) water flow regulation (c) carbon storage (d) soil retention (e) soil accumulation.

When all five ecosystem services were considered in the same plan, the best solution for Scenario 1 (20% target) showed that 24% of all catchments were needed to meet the targets. These catchments were found all over the study area except in the western parts (Figure 5.2). About 8% of all catchments had an irreplaceability of >0.5 in this scenario. The figures were higher for Scenario 2 (40%target) where 40% of catchments were needed to meet the specified targets and about 40% of the catchments had irreplaceability values greater than 0.5 (Figure 5.3a and b). The map of the best solution for the 60 percent target in Scenario 3 selected an even higher number of catchments (57%). A large number of catchments (>90%) had an irreplaceability values >0.5, but less than two percent had a value > 0.75.

The percentage number of catchments selected to meet targets when aligning biodiversity priorities with those of ecosystem services in Scenario 4 (Integrated plan) was slightly lower that than obtained in Scenario 2 (40%) which had the same parameters as Scenario 4 (except that biodiversity was not considered). The irreplaceability results from Scenario 4 (integrated plan) showed fewer catchments (< 10%) with irreplaceability values > 0.5 were recorded. However, the integrated plan had a relatively higher percentage of catchments with irreplaceability value > 0.75 (Figure 5.4a). These catchments were mostly in KZN, FS and NW (Figure 5.4b).

As expected, the percentage area selected as ecosystem service priorities increased with targets levels. Overlap between ecosystem service priorities and freshwater and terrestrial priorities were moderate to high (Table 5.1). There was at least a 40% overlap between catchments selected in all four scenarios with fresh water and terrestrial biodiversity priorities. The integrated plan had the highest overlap with freshwater biodiversity priority map.

Table 5.1: Percentage proportional overlap between ecosystem services and biodiversity priorities.

	Biodiversity priority	
Ecosystem service priorities	**Fresh water**	**Terrestrial**
Scenario 1 (20% target)	41	55
Scenario 2 (40% target)	46	49
Scenario 3 (60% target)	60	56
Scenario 4 (Integrated plan)	92	54

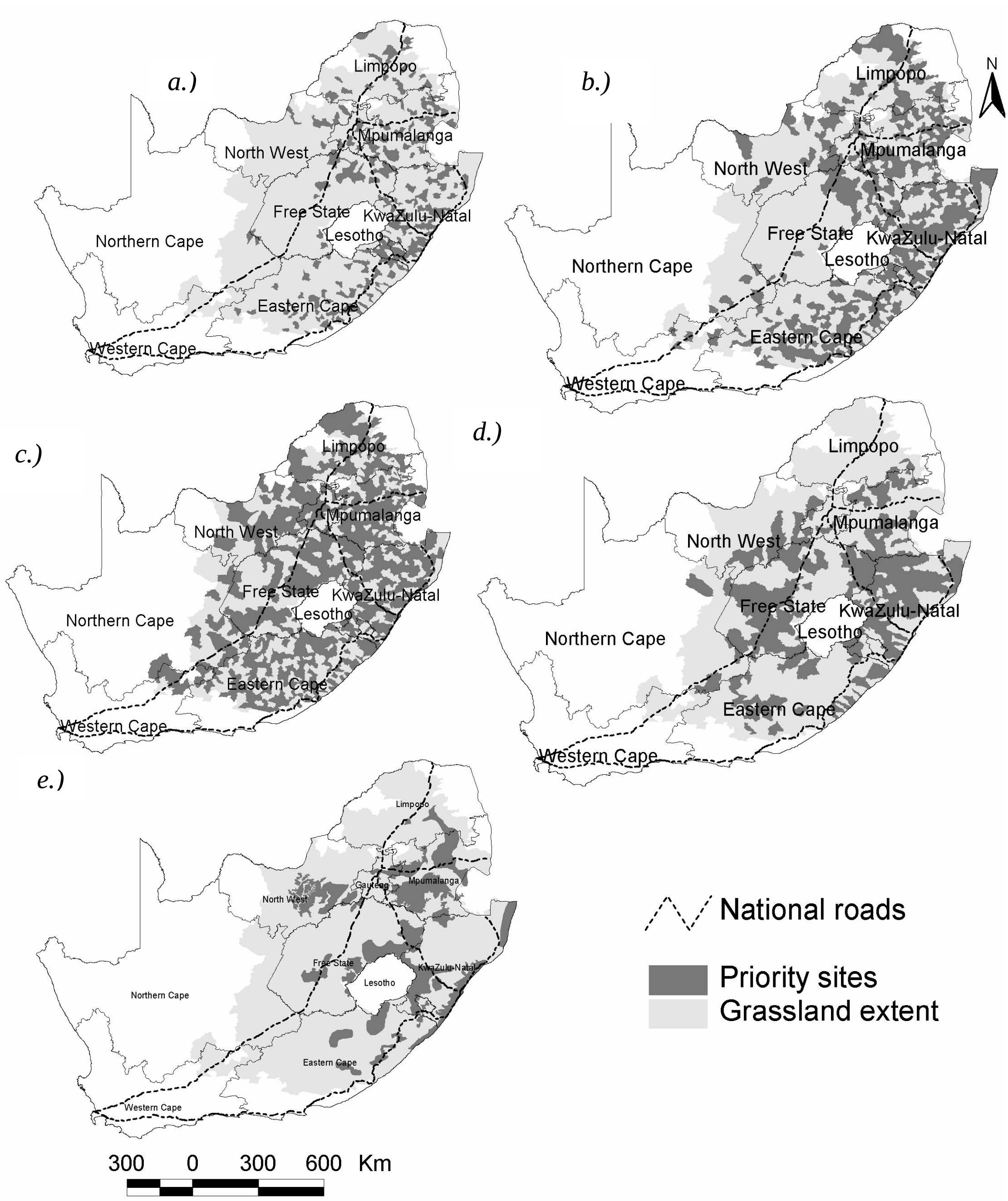
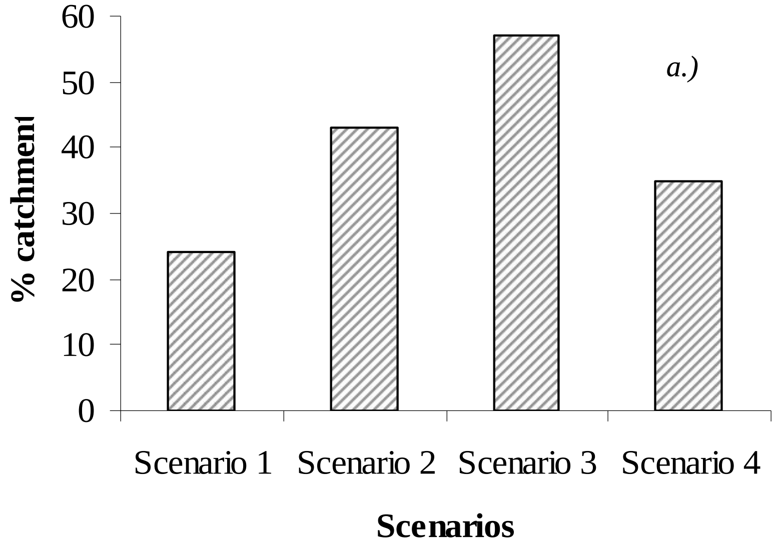


Figure 5.2: Priority areas for ecosystem services (based on best solution), rivers biodiversity and terrestrial biodiversity (a) scenario 1 (20% target) (b.) scenario 2(40 % target) (c) scenario 3 (60 % target) (d) fresh water biodiversity priorities (e) terrestrial biodiversity priorities.



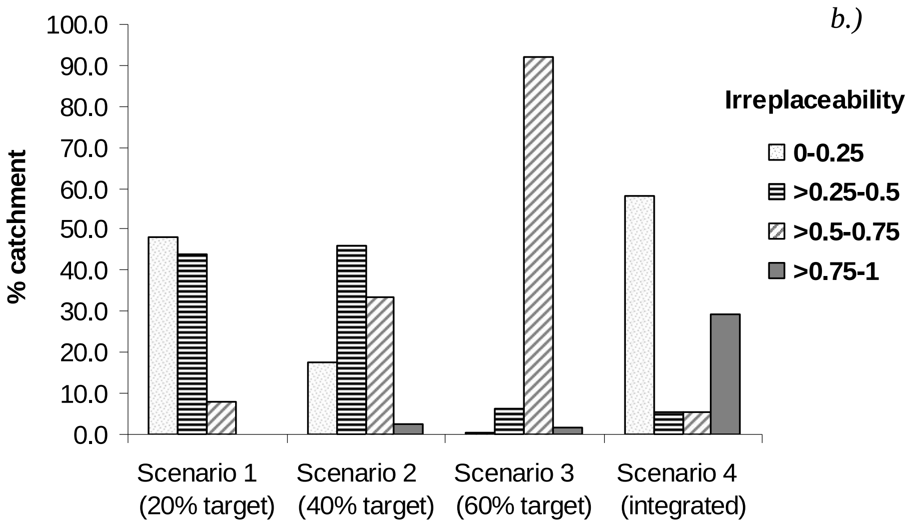


Figure 5.3: (a) Percentage of catchments selected as best solution for each scenario (b) percentage of catchments selected for each category of irreplaceability values.

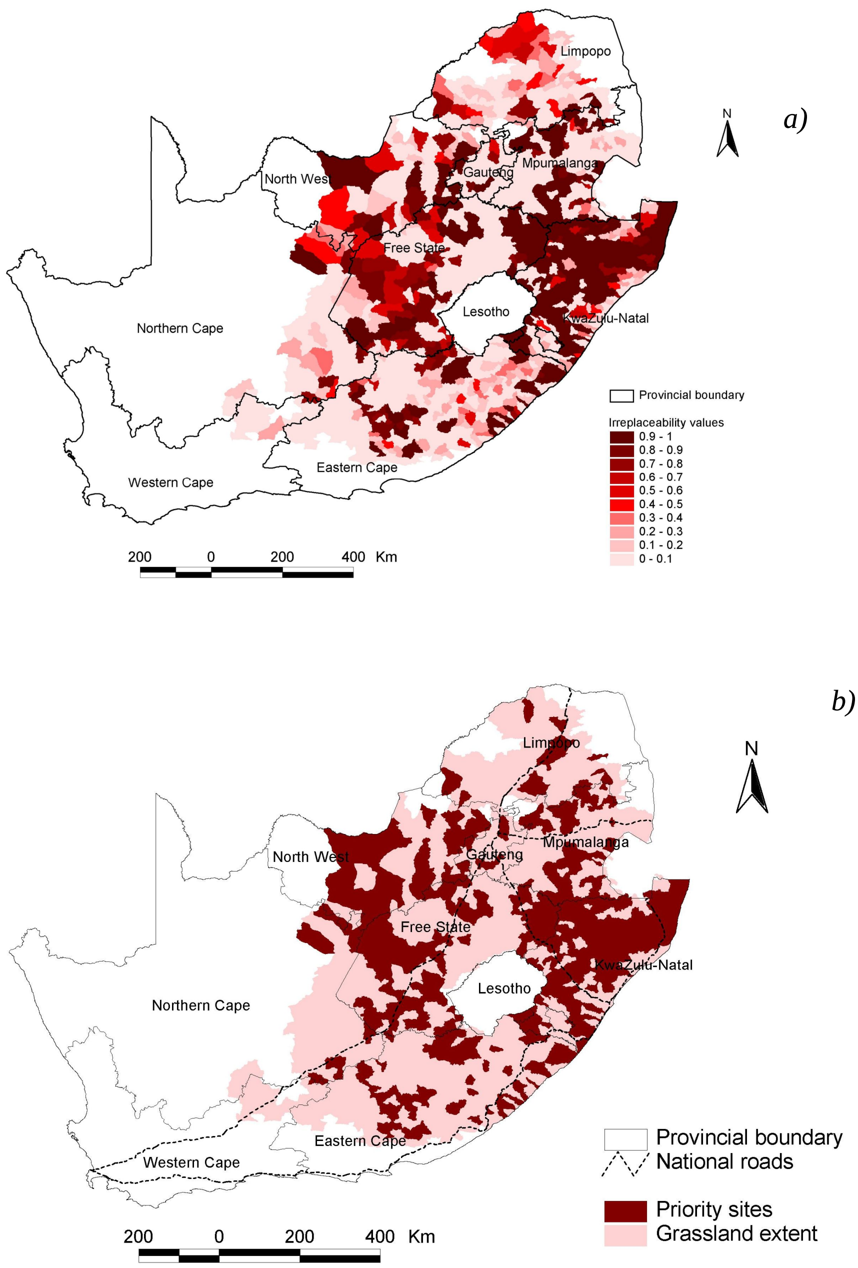


Figure 5.4: Priority maps for an integrated plan that includes both ecosystem services and biodiversity (a) irreplaceability (b) best solution.

Parts of the areas selected to meet various targets for ecosystem services in the four scenarios, have been degraded or transformed by various land uses. At least 25% of the areas selected in all four scenarios has been degraded or transformed (Figure 5.5). Scenario 1 had a relatively higher transformation and degradation (30%) levels compared to Scenario 2, 3 and 4. In general, the level of transformation and degradation found in these priority areas were similar to that of the grassland biome.

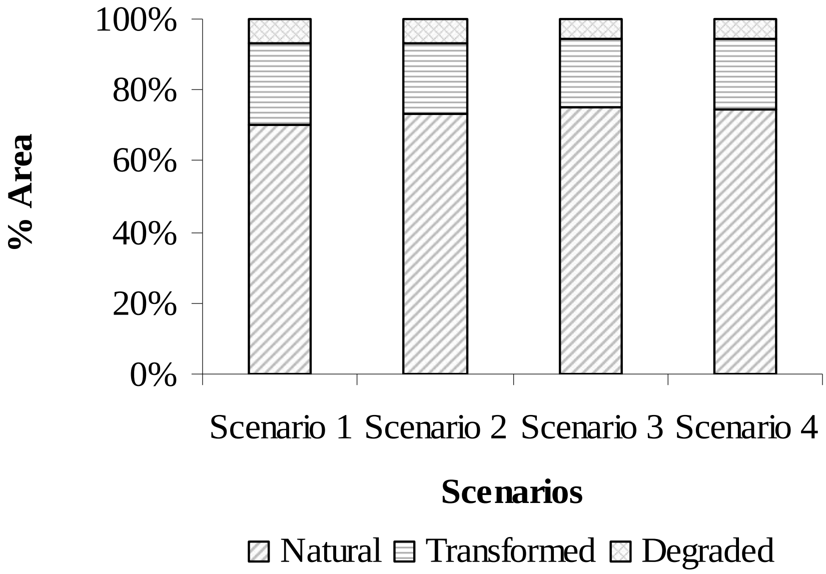


Figure 5.5: Transformation status of area selected to meet various targets for ecosystem services and biodiversity.

5.4 Discussion

This study set out to address the challenge of identifying priorities for ecosystem services in South Africa's grassland biome. When individual ecosystem services where planned for, MARXAN met targets for each service. The results showed the importance of catchments in the eastern KZN. These results suggest that focusing conservation efforts in catchments in the KZN region has a good potential of ensuring the delivery of multiple ecosystem services. However, specific catchments will need to be targeted to yield specific ecosystem services. Figure 5.1(a-e) shows which catchments will be needed to meet 40% targets for carbon and the water services and 50% targets for the soil services. Any effort to conserve the service of carbon storage proves to be more challenging, with selected catchments scattered all over the study area. Planning for individual services gives the opportunity of using management strategies specific to a particular service. However, a plan considering all services is needed to minimise the area requiring management for all services.

When all five ecosystem services were considered in the same plan, the number of catchments with an irreplaceability value >0.5 increased with the target levels. The percentage of catchments selected for the best solution also increased with target levels. This implies that stakeholders must decide what levels of targets are needed for management and tradeoffs with other forms of land use will have to be explicitly considered. This could be based on what is needed to ensure sustainability of service delivery or the resources available for implementation. It also highlights the importance of the full assessment as laid out in Cowling et al. (2008). A social and valuation assessment would be of great relevance to the setting of ecosystem service targets, identifying which services, how much is needed and who should pay. Management strategies for the best solution for any of the target levels in Scenario 1-3 might be challenging because the catchments selected were scattered all over the study area. It seems that an approach geared towards the entire landscape may be more appropriate.

The second objective of this study was to align ecosystem service objectives with those for biodiversity. About 30% of catchments selected in the integrated plan were highly irreplaceable with values > 0.75. The best solution in the integrated plan (Scenario 4) mirrored the fresh water priorities. This could be due to the way the cost layer was derived. While the scores for habitat and species were scaled from 0-100, those for the freshwater priorities were the same (100), making catchments selected as freshwater priority more important in the analysis. Catchments that were selected for freshwater biodiversity were therefore more likely to be selected especially if they were also rich in biodiversity (species or habitat) and if they had a small catchment area. The number of catchments selected for the best solution from the integrated plan was similar to that of Scenario 2 although slightly lower. These results indicate that there is some level of overlap between catchments selected for services and those for biodiversity. This was evident in the overlap analysis where more than 40% overlap was observed between ecosystem service priorities and biodiversity priorities.

Overlap of biodiversity priorities and ecosystem services have generally showed a moderate to high overlap (chapter 4; Turner et al., 2007). Chan et al. (2006) has also reported a generally positive, but weak correlation between the two. Studies that have reported moderate to high overlap between biodiversity and ecosystem services such as this one, have mostly considered only a few services. A full range of services needs to be investigated with many biodiversity features to draw any conclusions. Naidoo et al. (2008) concluded that conservation priorities aimed solely at biodiversity may not conserve optimal levels of ecosystem services, and vice versa and concordance between the two cannot be assumed. Nonetheless, the moderate levels of overlap found in this study and in chapter 4 of this thesis, coupled with findings from other studies (Tuner et al., 2007) suggest that any plan for ecosystem service and biodiversity should ultimately aim at meeting targets for both.

Few studies have integrated biodiversity into ecosystem services planning as was done in this study. It seems there could be more benefits from including biodiversity into an ecosystem service plans than vice versa. First, Naidoo et al. (2006) found that optimising for individual ecosystem services captured as many species (only 22-35%) as did optimising for species themselves (which was no better than randomly selecting planning units). The integrated plan in this study had a relatively moderate overlap with the terrestrial biodiversity priorities and a high overlap with the freshwater priorities. Irreplaceability results also suggest that there may be more flexibility as to which catchments get selected to meet ecosystem services targets and it makes sense to align biodiversity priorities with ecosystem service priorities. Second, approaching conservation from an ecosystem service point of view increases the potential for securing funding and boosting implementation. Funds for conservation can be generated through payments for ecosystem service (PES) programmes (Naidoo and Ricketts, 2006; Turner et al., 2007). Carbon can be sold in a carbon market to generate income for conservation. PES programmes are beginning to gain momentum in South Africa with water and carbon services having the most potential as tradable commodities (Blignaut et al., 2008; Turpie et al., 2008).

The last objective of this study was to quantify the levels of transformation of different ecosystem service priorities. It was found that more than 30% of the areas selected in all four scenarios (best solution) have either been transformed or degraded. The targets met in this study are only potential amounts, rather than actual amounts of ecosystem service production due to the loss of ecosystem services in transformed and degraded areas. Although, these results are similar to the 30% degradation levels in the grassland biome, they are quite high compared to about 20% for all of South Africa (Rouget et al., 2004). These high transformation and degradation levels can be related to the vulnerability of the grasslands to cropping and other land uses. A study by Olsen and Dinerstein (1998) listed the South African grasslands as critically endangered. Furthermore, this biome contains more vulnerable, endangered or critically endangered vegetation types combined than any other biome in the country (Reyers et al., 2007). The status of ecosystem services mirrors the biodiversity status and this study further highlights the urgent need for action in the grassland biome.

Despite their biodiversity and ecosystem service value, coupled with the considerable threats to their biodiversity, grasslands are one of the least conserved biomes in the world. Globally, just over 7% of the grasslands fall within protected areas, with the world's least conserved biome being temperate grasslands (< 0.69% protected; Henwood, 1998). In South Africa, only 1.9% of the grassland is in formal protected areas compared to an average of 9.8% for all biomes (Rouget et al., 2004). However, there is an emerging interest in conservation concerns for the grasslands both globally and locally (Bohensky et al., 2004). The National Grasslands Biodiversity Program (www.sanbi.org) is an example of this increased interest. Its inclusion of ecosystem services, as well as its focus on mainstreaming biodiversity into the production sectors of the grasslands is a new focus for conservation in the country. This study highlights the role that systematic approaches to ecosystem service mapping and planning can play in these projects, ensuring transparent, defensible and negotiable priority areas for ecosystem services.

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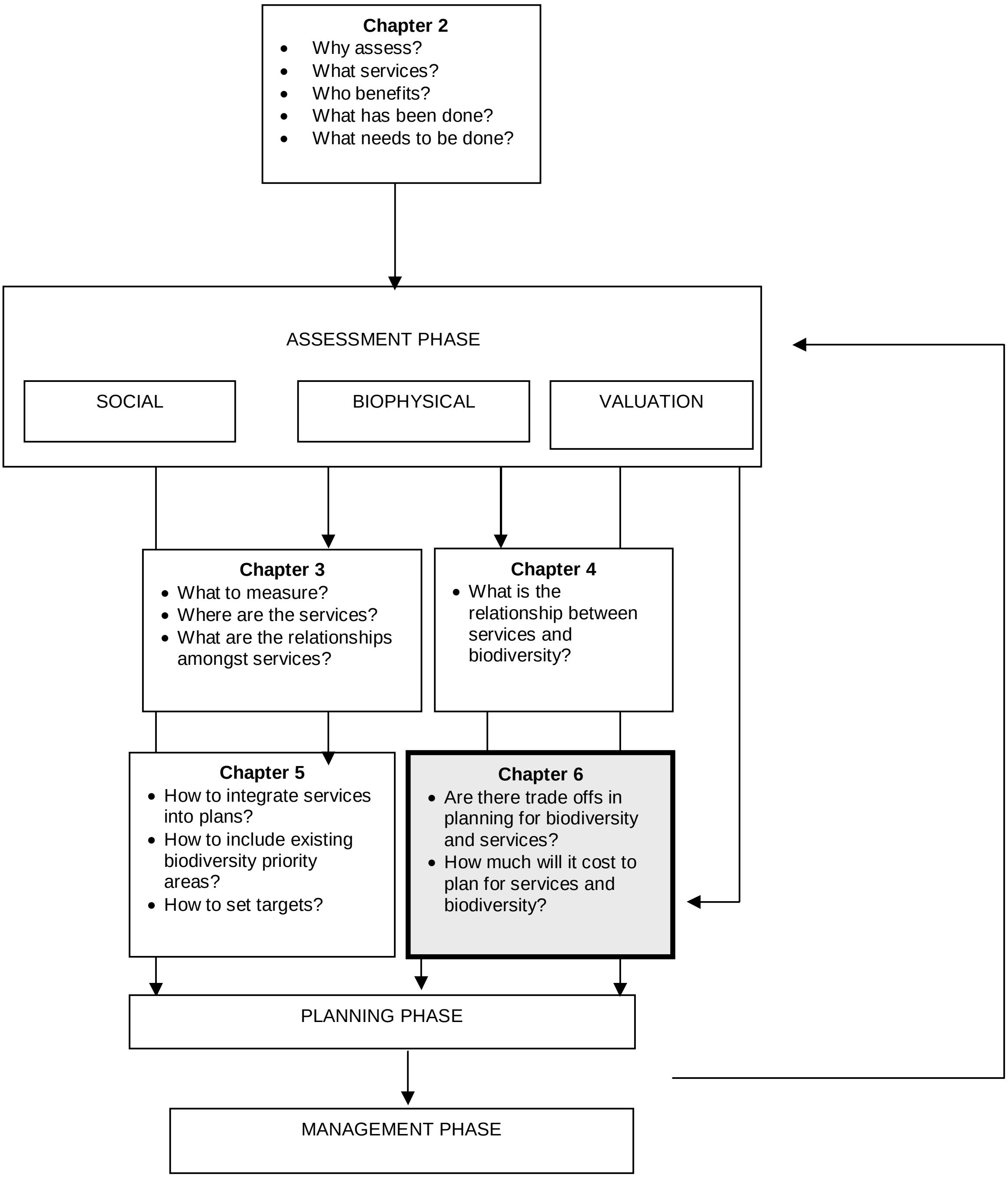
Chapter 2
Why assess?
What services?
Who benefits?
What has been done?
What needs to be done?
ASSESSMENT PHASE
SOCIAL
BIOPHYSICAL
VALUATION
Chapter 3
What to measure?
Where are the services?
What are the relationships amongst services?
Chapter 4
What is the relationship between services and biodiversity?
Chapter 5
How to integrate services into plans?
How to include existing biodiversity priority areas?
How to set targets?
Chapter 6
Are there trade offs in planning for biodiversity and services?
How much will it cost to plan for services and biodiversity?
PLANNING PHASE
MANAGEMENT PHASE

CHAPTER 6. SAFEGUARDING BIODIVERSITY AND ECOSYSTEM SERVICES IN SOUTH AFRICA'S LITTLE KAROO: SYNERGIES AND TRADE-OFFS

ABSTRACT

The decline of ecosystems and widespread degradation of ecosystem services places an urgent need for developing conservation strategies to safeguard them. Ecosystem services are often used to justify biodiversity conservation and there are calls to explicitly account for them in biodiversity conservation interventions. Payments for ecosystem services can be used to generate money for biodiversity conservation where there are synergies. While the benefits of safeguarding both are clear, the trade-offs in the costs of conserving biodiversity and ecosystem services are not well understood. In this study I set out to investigate the synergies and trade-offs in conserving ecosystem services and biodiversity in South Africa's Little Karoo. Three ecosystem services (carbon storage, water recharge and fodder provision) were mapped. I investigate the amount of each ecosystem service captured incidentally by a plan geared towards conserving biodiversity, where the objective was to meet variable targets for 369 vegetation types. I then examine the cost implications of adding targets for ecosystem services, and explore tradeoffs between the two for a fixed budget. I used simulated annealing within MARXAN version 1.8.2 to carry out the analysis. At least 30% of each ecosystem service was captured incidentally when 100% of biodiversity targets were met. Whilst holding the budget constant, I was able to increase the amount of ecosystem services captured by at least 20% for all three services. The opportunity cost (in terms of forgone production) of safeguarding 100% of the biodiversity targets was about USD $ 500M. The net present value of the land that met this target was about USD$8.3 billion. When biodiversity targets were reduced by 8%, an extra 40% of fodder provision and water recharge, and 58% of carbon are captured for the same budget. These results have implications for implementation. They show that with a small increase in budget, or a small decrease in biodiversity targets, we can achieve substantial gains for the conservation of both ecosystem services and biodiversity.

Keywords: Conservation planning, ecosystem services, implementation, cost, carbon, water, fodder.

6.1 Introduction

Ecosystem services have been used for many years by conservation biologists as a rationale for conserving biodiversity (Armsworth et al., 2007), but are rarely explicitly considered in conservation plans (Singh, 2002; chapter 2; see Chan et al., 2006; Nelson et al., 2008 for notable exceptions). Ecosystem services are the benefits that humans derive from natural systems, which in turn rely on a certain level of biological infrastructure (Myers, 1996; Balvanera et al., 2001). Despite conservation efforts, both biodiversity and ecosystem services are declining (MA, 2005; Conrad et al., 2006). Reports on the continuous degradation and unsustainable use of ecosystems services around the world highlight the urgent need to develop strategies to safeguard ecosystem services as well as biodiversity (Balvanera et al., 2001; Kremen and Ostfeld, 2005; van Jaarsveld et al., 2005; Chan et al., 2006). Responses to this urgency include the emergence of new initiatives on ecosystem service planning and management (e.g. The Natural Capital Project (http://www.naturalcapitalproject.org) and the Valuing the Arc project (Fisher and Turner, 2008)).

There are many challenges in developing plans to direct efforts on the conservation and sustainable use of ecosystem services (chapter 2; Kremen, 2005). Planning approaches developed for conserving biodiversity are not necessarily suitable for targeting ecosystem services. First, there are many conceptual issues around the definition and classification of ecosystem services (Wallace, 2007, 2008; Costanza, 2008; Fisher and Turner, 2008). Consequently, any ecosystem service assessment demands a biophysical and social assessment in order to understand the social-ecological system and the values and perceptions of stakeholders (Cowling et al., 2008). Second, data on the functioning and spatial distribution of ecosystem services are limited and there is no universally accepted method for quantifying and mapping ecosystem services (Naidoo et al., 2008). Third, safeguarding ecosystem services demands the involvement of multidisciplinary teams and may involve a variety of stakeholders and implementation agencies. Lastly, safeguarding biodiversity is already very expensive; adding additional ecosystem service objectives may compound the problem if the two objectives are not spatially congruent (Frazee et al., 2003). Despite these challenges joint strategies to safeguard both biodiversity and ecosystem services have been proposed and opportunities for such synergies have been sought on global (Turner et al., 2007; Naidoo et al., 2008), regional (van Jaarsveld et al., 2005) and local scales (Chan et al., 2006). Such proposals have typically been from the perspective of using the conservation of ecosystem services to justify the conservation of biodiversity where necessary (Armsworth et al., 2007, but see McCauley, 2006).

Although there have been considerable advances in approaches for systematically identifying priority areas for biodiversity conservation (Margules and Pressey, 2000; Sarkar et al., 2006), implementation of the resulting biodiversity plans remains a challenge. Implementation is a crucial phase of conservation planning, but has often been neglected in previous efforts (Knight et al., 2006). This is partly because implementation success depends on people's value systems coupled with the availability of funds (Freyfogle and Newton, 2002; Pierce et al., 2005). Ecosystem services are closely linked to human values and to particular beneficiaries. Payments for ecosystem services (PES) could generate money that may be used for conservation where synergies exist and organizations could work together to improve implementation success (Chan et al., 2006; Naidoo and Ricketts, 2006; Turner et al., 2007). A recent advance in priority setting is the inclusion of economic costs (e.g. land acquisition, management, compensation, return on investment) from the start of the priority-setting process (Wilson et al., 2006; Murdoch et al., 2007) as a means of reducing costs of implementation and producing effective conservation actions (see Naidoo et al., 2006 for review). The inclusion of ecosystem services in conservation planning should also help to improve the relevancy and ease of implementation of conservation priorities.

While the potential benefit from an integrated plan for ecosystem services and biodiversity seems logical, the trade-offs and costs of safeguarding both simultaneously are still unclear. Few studies have investigated the trade-offs associated with trying to safeguard both ecosystem services and biodiversity (Chan et al., 2006; Naidoo et al., 2008; Nelson et al., 2008). Chan et al., (2006) were among the first - they evaluated the additional area required to meet ecosystem service targets over and above biodiversity targets. In another study, Naidoo and Ricketts (2006) mapped the economic costs and benefits (including ecosystem services) of conservation reserves in eastern Paraguay. Neither of these studies investigated the integration of both objectives simultaneously in a single conservation assessment.

The aim of this study is to evaluate the synergies, trade-offs and economic costs of safeguarding ecosystem services and biodiversity by increasing the area of land managed for conservation in the Little Karoo of South Africa. Specifically I (a) evaluate the amount of ecosystem services captured incidentally by a conservation plan focused on biodiversity to achieve nationally-specified targets; (b) explore the opportunity costs for biodiversity conservation when seeking to conserve both biodiversity and ecosystem services for a fixed budget and (c) evaluate the cost increase when targets are fixed for biodiversity and targets for ecosystem services are gradually increased.

The major strategy for conservation planning in South Africa is to plan for sustainable land management across the entire landscape instead of focusing only on reserves (Driver et al., 2003). This approach demands a mixture of on- and off-reserve management often with different cost implications. The former involves land acquisition and reserve management while the latter mostly requires engaging with landowners for sustainable use of their land, and often providing payments or incentives (Pence et al., 2003). Off-reserve management improves landscape sustainability, and is essential in South Africa where over 80% of land is privately owned and not readily available for reservation (Frazee et al., 2003). In the study area, about 20% of the land is already protected and hence the current conservation focus is off-reserve management. An important off-reserve management strategy is through financial incentives such as compensation for forgone income or opportunity costs (Polasky et al., 2001; Pence et al., 2003; Naidoo and Ricketts, 2006). Therefore I used opportunity cost data in this study.

The approach used in this study improves on existing initiatives because I explore the spatial relationship between safeguarding biodiversity and ecosystem services in a conservation planning framework that includes economic costs and trade-offs between the two objectives. Specific amounts of ecosystem services and biodiversity are targeted rather than evaluating the overlap priority maps for each kind of conservation feature (biodiversity and ecosystem services) feature separately as in previous studies. I include a range of ecosystem services (water recharge, carbon storage and fodder provision) and one biodiversity feature (vegetation type). This study has the potential to communicate synergies and trade-offs between biodiversity and ecosystem service management.

6.2 Methods

6.2.1 Study Area

The Little Karoo lies within the transitional winter and summer rainfall zone of South Africa (Tyson, 1986) and covers about 19 000km^2. The region is semi-arid, with an uneven spatial distribution of rainfall, ranging from as little as 20 mm/yr in the west to over 1000 mm/yr in the southern mountains (Lynch, 2004; Le Maitre et al., 2008). The landscape consists of a range of mountains and valleys up to 50km wide and 200km long (Watkeys, 1999). The soils are deep on the valley floor and fine textured but elsewhere the soils are generally shallow (<300mm). The study area includes components of three recognized global biodiversity hotspots: the Succulent Karoo, the Cape Floristic Region, and Maputoland-Pondoland-Albany (Mittermeier et al., 2005).

The economy of the Little Karoo is dominated by agricultural production which is linked to and strongly dependent on limited water supplies (surface and ground) for irrigated agriculture and range-fed livestock (O'Farrell et al., 2008). A variety of fruits, vegetable, and cereal crops are grown in the study area, with lucerne (for livestock feed) being the dominant cultivated crop. Grazing is also widespread, with ostriches being the dominant livestock type. Irrigated crops use 90% of the water captured in the region and the current demand exceeds the estimated sustainable yield of the river systems of the Little Karoo by a substantial margin (Le Maitre and O'Farrell, 2008).

The Little Karoo is currently the focus of a major conservation initiative called the "Gouritz Initiative" (www.gouritz.com) aimed largely at implementing conservation actions outside protected areas. The Gouritz Initiative (GI) is a partnership between various conservation planning programs for the region and numerous governmental and non-governmental stakeholders. The objective of the GI is to safeguard both biodiversity and ecosystem services in the study area.

6.2.2 Data description

6.2.2.1. Biodiversity

Vegetation types were used as a surrogate for the biodiversity in the study area. Vlok et al., (2005) provided a fine-resolution map (1:50,000) of a hierarchy of six biomes, 56 habitats types and 369 vegetation types. The 369 vegetation types were aggregated into 32 major types relevant to the agriculture and wildlife industry in the region by considering their physiognomy rather than the floristic component of the vegetation units (Vlok et al., 2005). No comprehensive fine-scale coverage of species point locality data exists for the study area. Biodiversity targets were developed for the Little Karoo vegetation types at the level of the habitat type, based on species turnover (Desmet and Cowling, 2004); these range from 16% to 34% of original extent. A land cover and degradation map of the study area was used to evaluate the amount of vegetation remaining in a pristine or moderately degraded condition (Thompson et al., 2005) to contribute to meeting biodiversity targets. Areas that are cultivated, urban or severely degraded were classified as transformed and not considered to contribute to biodiversity objectives. Restoration of severely degraded areas (at the scale that would be required to increase conservation value substantially) is not currently considered a viable option in this region, because of the low returns in investment in such initiatives (Herling et al., 2007). Since I did not consider restoration as a conservation action, I could not consider transformed areas.

6.2.2.2. Ecosystem services

I considered three ecosystem services in the study area: carbon storage, natural vegetation fodder provision (hereafter referred to as "fodder provision"), and water recharge. These services were selected for their importance to the economy in the study area. I do not differentiate between ecosystem services and ecosystem functions, but use the term ecosystem services to include all ecosystem attributes, entities and processes that provide actual or potential benefits to humans. I estimated the amount of each ecosystem service provided by each vegetation type under both pristine, degraded (moderate and severe in some cases) conditions as deduced from the land cover map. Ecosystem services generated in transformed and urbanised areas were ignored for carbon storage and fodder provision because these were negligible. Below I provide a brief description on how the ecosystems services were mapped.

Carbon storage: Carbon storage is the amount of carbon stored below or above the ground. The retention of this carbon has the potential to mitigate climate change impacts. Carbon storage in the region has been found to exceed 20kg/m^2 in intact thicket (Mills and Cowling, 2006). Experts used this information to estimate carbon storage potential for each of the 32 major habitat types in tonnes per hectares. I used these data to estimate carbon storage per planning unit (the building block of a reserve network) (1km^2 grids) for the entire study area. I assume that if an area is selected for conservation, the storage of this carbon will be secured.

Fodder provision: In South Africa, livestock carrying capacity is measured as number of hectares required per large stock unit (LSU) for sustainable grazing. The carrying capacity recommendation map of the Department of Agriculture was overlaid with the fine-scale vegetation map. The area (ha) per LSU required for sustainable fodder provision was calculated for each habitat type for pristine, moderately degraded and severely degraded conditions. This was then converted to the number of LSU per planning unit. It was assumed that if an area was selected for conservation, grazing rates would be reduced to sustainable levels, thus providing fodder for wildlife or livestock. This type of grazing is different from unsustainable grazing where land is overstocked with livestock.

Water recharge: Ground water is the main regulator of water flows in river systems. The most appropriate dataset for estimating these flows were gridded data on groundwater recharge (DWAF, 2005). Estimates of the amount of potentially available water were derived from data on rainfall amounts, lithology and chloride concentrations in groundwater, (DWAF, 2005) because high sodium chloride (salinity) concentrations make the water unfit for domestic use. Groundwater recharge was estimated for pristine, moderately degraded, and transformed areas separately. The ecosystem service was mapped as millions of m^3 of groundwater recharge per 1 km^2 grid cell. It was assumed that conservation in an area secured this provision of good quality groundwater.

6.2.2.3. Cost

In this study, I estimated the opportunity cost of conservation using data on modelled gross income, which I used to calculate the net present value of the land (Polasky et al., 2001). The cost of conserving each planning unit in perpetuity was equal to the amortised maximum gross income (based on farm gate prices) that the land owner could obtain by putting that planning unit into the most profitable land use. I assume farmers act rationally, choosing the land use that will maximise their economic benefits (although often only in the short term). I was unable to consider spatially variable production costs (as per Naidoo and Iwamura, 2007), and thus I effectively assume that these are homogenous over the study area. I used a discount rate of 6% to calculate the net present value (NPV) of the land (Wilson et al. 2007, Carwardine et al., 2008). The NPV represents the minimum cost that a rational landowner will accept as compensation for managing their land for conservation in a stewardship program. I calculated this value per hectare for cultivation of wheat, barley, lucerne, tobacco, deciduous fruits and vegetables. Since most uncultivated areas with potential for cultivation are near areas already cultivated, I determined potential cultivation areas by buffering existing cultivated areas by a 500m radius. Although cultivation could expand beyond 500m of cultivated areas, I choose this value to avoid an overestimation especially because slope was not considered, plus it's been shown that most lowland area in the Cape Floristic Region has already been cultivated. I used farm gate prices to calculate gross income per hectare of cultivating fruits and vegetable. Gross income varied over the study region. I used the mean value for each type in this analysis. The maximum value of potential or actual revenue generated from cultivation was summarised per planning unit.

I calculated gross income from grazing at unsustainable levels by doubling the number of LSU/ha for sustainable levels of fodder provision. I assumed that at this stocking rate pristine vegetation will become severely degraded in a maximum of 20 years. This has been the case in the succulent Karoo where recommended stocking rates more than 20 years ago were twice as much as those of today, which has significantly contributed to present levels of degradation (Todd and Hoffman, 2001; Anderson and Hoffman, 2007). The number of LSU per planning unit was calculated at sustainable and unsustainable levels, and so was the gross income based on farm gate prices. Gross income per planning unit was compared with those from other land uses and the maximum income was assigned to each planning unit.

6.2.3 Analysis

I used simulated annealing within MARXAN vs1.8.2, which selects sets of conservation areas that meet targets for conservation features (usually types of biodiversity) at a minimal cost (Possingham et al., 2000). All data (e.g. extent of each vegetation type, carbon storage and opportunity cost) were summarised at the level of the planning unit (1km^2 equal sized, 19357 in total). MARXAN selects multiple sets of alternative networks, all of which are near optimal at achieving the conservation objective. A species penalty factor (SPF) determines the importance of meeting targets – higher penalties for features increase the likelihood of the target being met.

Four scenarios were designed to evaluate the cost of different conservation strategies for safeguarding biodiversity and/or ecosystem services in the Little Karoo (see below). A zero cost was assigned to any planning unit classified as protected and these were selected in every scenario. At least 100 runs with 1,000,000 iterations each were used for each analysis. These parameters were chosen in consultation with the authors and frequent users of MARXAN. I used 100 runs instead of 1000 in chapter five because the analysis was intense and some had to be done several times.

Scenario 1: Biodiversity targets only, no ecosystem services target, no budget

The objective of this scenario was to assess the amount of an ecosystem service captured in areas selected to meet biodiversity targets most efficiently. Different biodiversity targets were used. I started with 10% of the biodiversity targets (which ranged from 16% to 34% of the areas per each vegetation type) and increased this percentage each time by 5% up to 100%. I placed a high penalty on not meeting a biodiversity target so that each time all targets were met. I estimated both the opportunity cost of achieving the targets and the amount of ecosystem service captured incidentally by the resulting conservation area network for each biodiversity target level. Results were summarised using the best solution from MARXAN at each target level, which is the network that meets the targets at the least cost. I compared the amount of ecosystem services captured to a randomly drawn sample (equal to the number of planning unit selected to meet biodiversity targets at each target level). Each random sample had 100 iterations.

Scenario 2: Biodiversity and ecosystem services target with fixed budget

Here I investigate the difference between targeting and not targeting ecosystem services in a biodiversity conservation assessment under a range of biodiversity target levels from *Scenario 1*. I explored the ecosystem services that can be captured under different amounts of biodiversity targets when targets (from *Scenario 1*) for ecosystem services were explicitly sought – but I sought to meet the ecosystem service targets without increasing the budget (from *Scenario 1*) for the whole plan. The biodiversity targets and resultant cost (cost of meeting biodiversity targets of 10% to 100%) in *scenario 1* were held constant and ecosystem service targets were introduced and gradually increased to find the maximum ecosystem service captured for each target level for the same budget (e.g. how much more (compared to that met incidentally) ecosystem service can be captured when 10% of a biodiversity target are met for the same cost?). I achieved this by setting the SPF for biodiversity greater than that for ecosystem services, which was set at a value greater than 1 – to favour the meeting of biodiversity targets.

Scenario 3: Trading off biodiversity and ecosystem services for a fixed budget

The objective of this analysis was to evaluate the tradeoffs between biodiversity and ecosystem services by finding out how much ecosystem services can be captured by giving up some biodiversity targets. A cost threshold was set (at the cost of meeting 50% of biodiversity targets), and different ratios of biodiversity/ecosystem service conservation were explored. Meeting the full target for all vegetation types required a large part of the study area, allowing little flexibility in trading biodiversity for ecosystem services. I therefore chose the cost of meeting half of the biodiversity targets (8% to 16% for different vegetation types) as my budget, and explored the ratio of biodiversity/ecosystem service target met for this budget. *Scenario 1 or 2* told me how much of each ecosystem service is captured by meeting 50% of the biodiversity targets. I systematically increased this amount for each ecosystem service and calculated the number of biodiversity features whose targets were not met for the same budget. Each time, I assigned a higher SPF to the ecosystem service than the biodiversity features. The percentage of biodiversity features whose targets were met was estimated at each incremental increase in percentage of ecosystem service targets.

Scenario 4: Biodiversity and ecosystem services with flexible cost.

The objective here was to assess the opportunity cost of increasing targets for ecosystem services. I held biodiversity targets constant (at half the original targets) and systematically increased targets for each ecosystem services and calculated the cost increase of the resulting conservation area network. The starting target for each ecosystem service was the amount captured incidentally while planning for biodiversity alone and meeting 50% of the biodiversity target. I had the same SPF for both biodiversity and ecosystem services. I used the best solution from MARXAN to summarise results.

6.3 Results

Ecosystem service mapping

The total amount of carbon stored in the study area was about 8.3*107 tons of carbon. The valley thicket with spekboom *(Portulacaria A.)* had the highest amount of carbon stored per hectare (250 tonnes). This habitat type also had the smallest number of hectares per LSU favouring fodder provision. The study area could sustainably provide fodder for about 21585 LSU (wildlife or livestock). The total amount of ground water recharge for the study area was 3.8*108 Mil m^3. Water recharge is mostly a function of habitat, soils, rainfall and slope. Carbon storage and fodder provision are driven more by habitat types.

Opportunity costs across the landscape

There was a high degree of cost variability throughout the study site (Figure 6.1). The maximum gross income that could be made by putting any planning unit into any land use varied from R0-R3 424 000 ($489 000; exchange rate USD$1=R7). The net present value (NPV) of planning units ranged from R0 to R57 062 000 ($8 152 000). The gross income from grazing was generally lower than that for cultivation. The most profitable land use was the cultivation of deciduous fruits. The cultivation of vegetables, wheat and citrus fruits also produced more gains than other land uses. The average maximum revenue that could be generated for any land use per planning unit was R766 000 ($109 000). The average NPV per planning unit was R 12 762 000 ($1 823 000). Most of the high value lands were concentrated in the low-lying eastern parts of the study area.

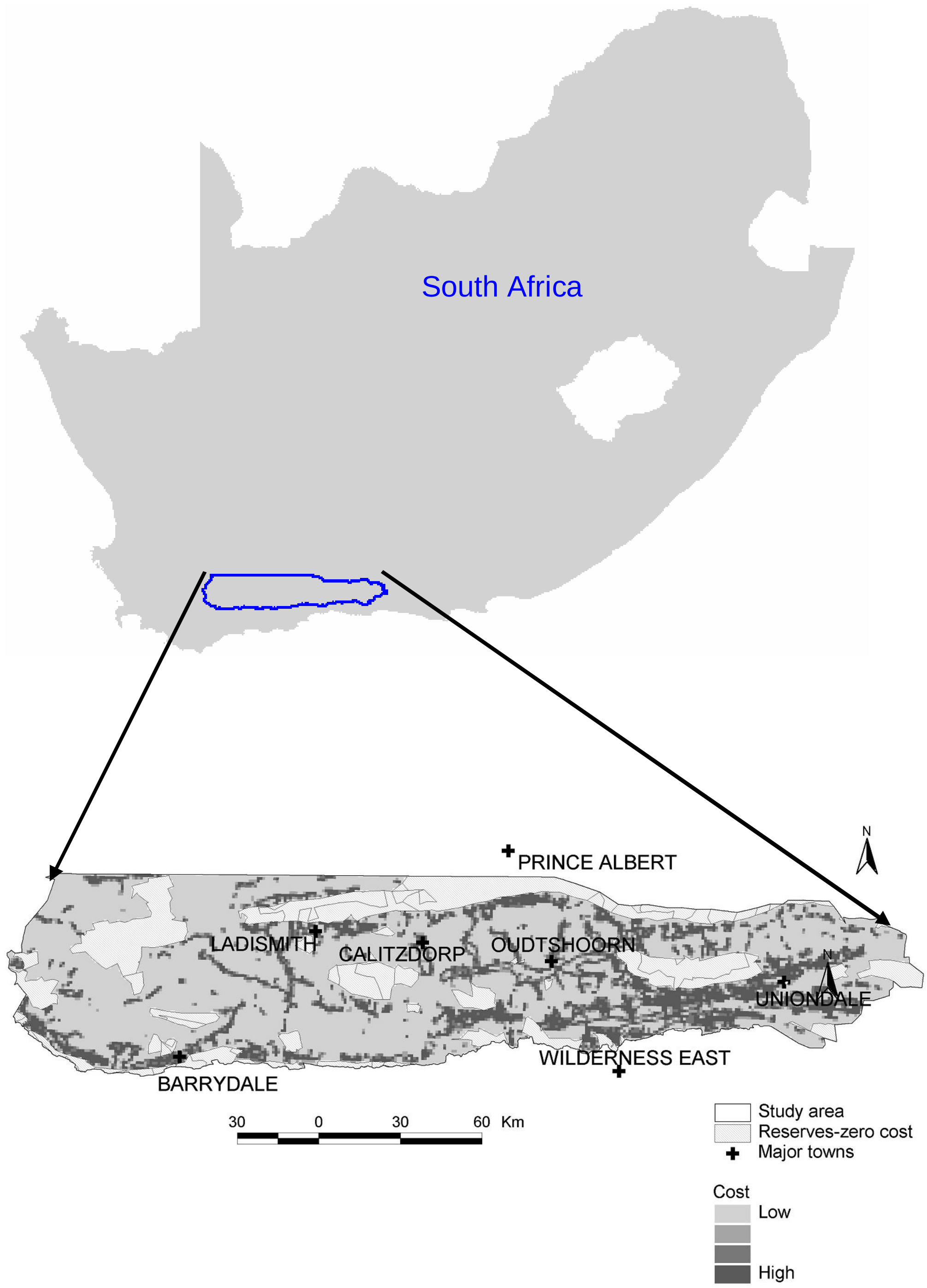

Figure 6.1: Map of study area with protected areas and cost of planning units.

The opportunity cost of reaching 100% of the biodiversity targets was about US $ 500M per annum. The NPV of the land that met this target was about US$8.3 billion. When only 50% of the biodiversity targets were met, the opportunity cost of safeguarding biodiversity dropped to about US$200M with the NPV of the land required also dropping to about US$3.2 billion.

Scenario 1: Biodiversity targets only, no ecosystem services target, no budget

When the full biodiversity targets were met, approximately 37% of all carbon stored in the study area, 45% of all LSU and 57% water recharge were captured incidentally. Meeting 50% of the target captured 23% of carbon, 32% fodder and 48% water recharge. I observed a roughly linear increase in both the amount of ecosystem service captured incidentally and cost (and the area requiring conservation) as I increased biodiversity targets (Figure 6.2a-c). However, the amount of LSU captured incidentally was not significantly different from that captured by the random sample. Nevertheless, the amount of carbon captured incidentally was always lower than that captured by random, while water recharge was significantly higher than the random sample.

Scenario 2: Trading off biodiversity and ecosystem services for a fixed budget

The amount of ecosystem services captured at no additional cost when explicitly targeting ecosystem services increased by at least 20% for water and 30% for carbon and fodder provision (Figure 6.2a-c). While the overall cost was held constant, larger areas were selected when both objectives were sought (Figure 6.3a-d). The large variation in the cost of planning units in the study region, allowed MARXAN to trade expensive planning units selected for biodiversity only, with a greater number of cheaper ones that contributed to both biodiversity and ecosystem service objectives. For example, a reserve network aimed at meeting biodiversity targets while including 37% of carbon storage was 1.5 times larger than that for biodiversity only, but had the same opportunity cost. Although these two conservation area networks shared about 65% of the planning units, 9% of the planning units selected for the "biodiversity-only" network and not selected for the "integrated" network were very expensive. The difference between the planning units selected for the biodiversity only network and for both biodiversity and ecosystem service network was greatest for carbon compared to the other two services.

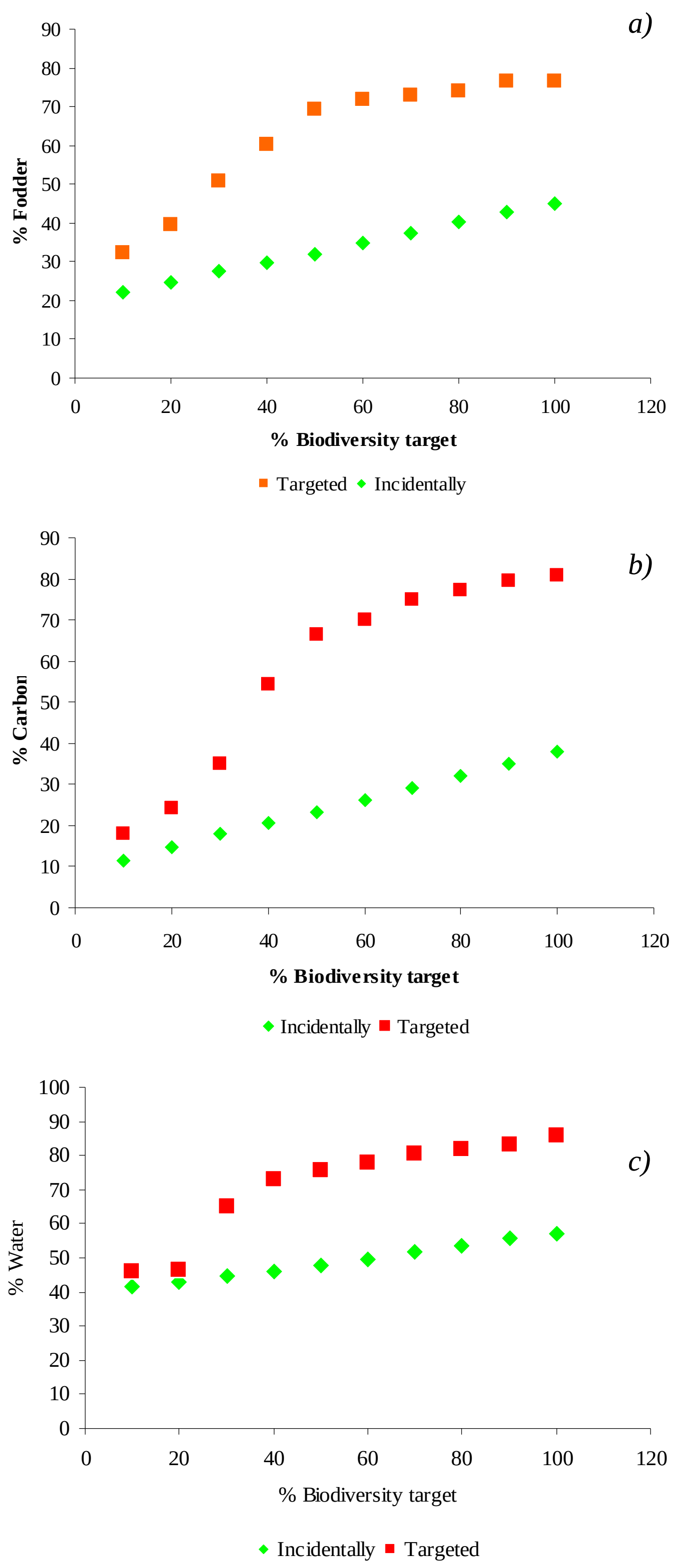

Figure 6.2: Percentage increase in ecosystem service captured by increasing biodiversity targets for two conservation plans, one of which targets ecosystem service (a) fodder (b) carbon (c) water recharge.

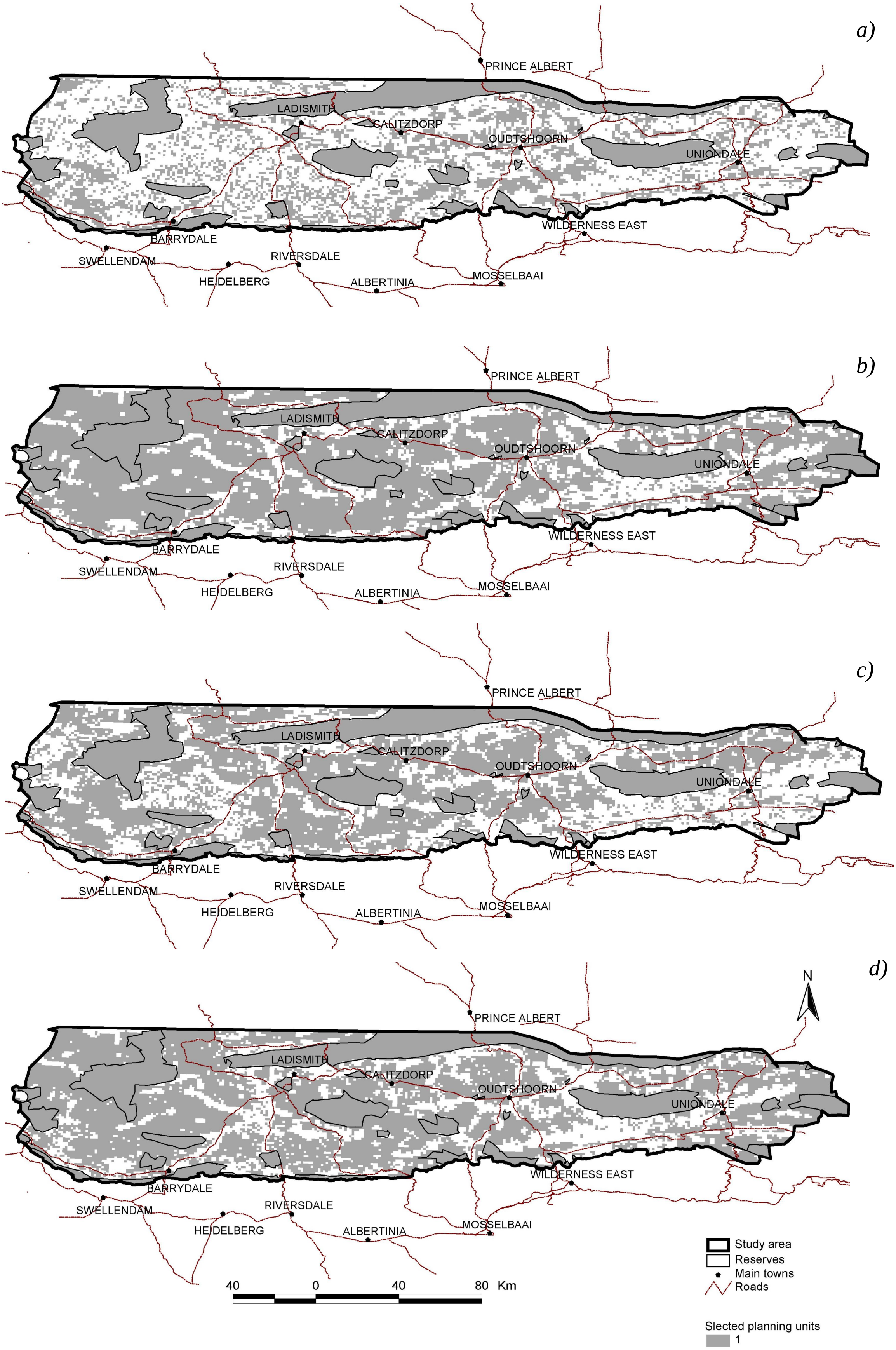

Figure 6.3: Map of study area showing conservation priorities when only biodiversity is targeted and when both biodiversity and various ecosystem services are targeted simultaneously (a) biodiversity only (b) biodiversity and fodder (c) biodiversity and carbon. (d) biodiversity and water recharge.

Scenario 3: Trading off biodiversity and ecosystem services for a fixed budget

Relinquishing small amounts of biodiversity targets resulted in big gains in ecosystem services. When targets for 8% of the biodiversity features are not met, an extra 40% of fodder provision and water recharge, and 58% of carbon are captured for the same budget. This indicates that there are areas which perform well in meeting biodiversity targets but not ecosystem services and vice versa.

Scenario 4: Biodiversity and ecosystem services with flexible cost.

Increasing targets for ecosystem services by about 30% did not significantly increase the opportunity cost of the network from the biodiversity-only amount (Figure 6.4). Although the cost did not increase, the percentage area required for conservation increased significantly (Figure 6.5). For example, a 10% increase in target for fodder provision did not increase the cost but resulted in a 10% increase in area. Beyond this amount, I could increase targets for carbon for a lower increase in cost than that for the other two services.

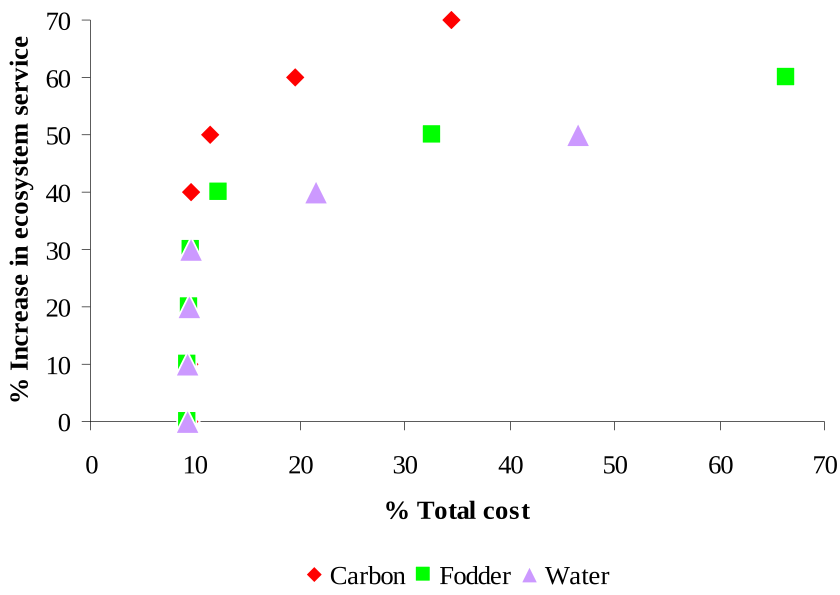

Figure 6.4: Cost of increasing targets for ecosystem services starting with a biodiversity plan that meets 50% of the targets for vegetation types at the minimum cost.

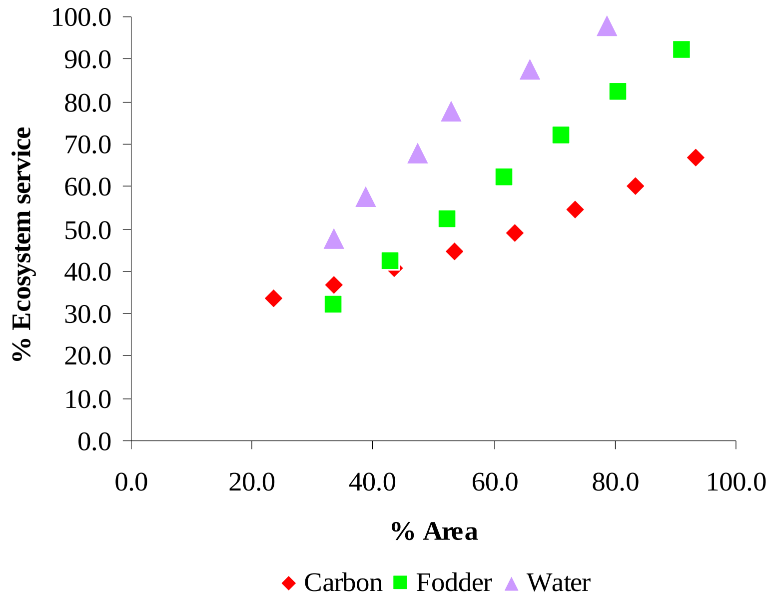

Figure 6.5: Percentage increase in area as ecosystem service targets increase.

6.4 Discussion

Can biodiversity conservation plans effectively incorporate ecosystem services?

The results obtained in this study consistently show that just by targeting ecosystem services explicitly, substantial amounts of ecosystem services can be secured in the study area alongside biodiversity at no additional cost. The reserve network for the full biodiversity target captured at least 10% more of each service when ecosystem services where targeted. In addition, giving up a small percentage of biodiversity targets resulted in big gains in ecosystem services for the same cost. Naidoo et al. (2008) found that by maximising species representation for a given area they captured 17-53% of ecosystem services. These results imply that although there may be areas where ecosystem services and biodiversity do not align, there is a possibility of meeting ecosystem service targets in a biodiversity plan. Nevertheless, ecosystem services will have to be explicitly targeted in such a plan. However, the large areas already reserved might have influenced the outputs in that when targets were included for ecosystem services, MARXAN selected additional areas at low cost with large benefits and often over large areas. These results are less likely to be found in a region with few areas under protection.

In general, findings from this study are in line with other studies that have found synergies between biodiversity priorities and ecosystem services and potential to combine the two objectives (Turner et al., 2007). Although Turner et al., (2007) found a generally high concordance between biodiversity priorities and ecosystem services across the globe, they also found low concordance between the two in the Succulent Karoo. This study contradicts their results in that I captured relatively high percentages (almost 40% of each) of ecosystem services in the study area when planning for biodiversity alone, which suggests high concordance between the two. However, the Succulent Karoo is generally low in productivity and hence ecosystem services which must have accounted for the low concordance found by Turner et al., (2007) at a global scale. Also, Turner et al (2007) used data from Costanza et al. (1997) which has been criticised by economics (Bockstael et al., 2000) for a number of methodology errors.

Conservation biologists are proposing a conservation strategy in which ecosystem services may be used to generate funds for biodiversity conservation where there are synergies. Our results suggest that this strategy will be useful, especially for carbon sequestration because the service has the potential to generate much income with the current carbon markets (see Naidoo and Ricketts, 2006). Although these results align with other previous studies, they should be interpreted with care as I have only focussed on three ecosystem services and a range of ecosystem services are needed to make appropriate conclusions.

What is the cost of conservation in the Little Karoo?

The opportunity cost of safeguarding 100% of our biodiversity target for the study area was very high compared to the management cost of a conservation plan for the Cape Floristic Region (Frazee et al., 2003). The amount reported in this study represents the opportunity cost of conservation, or the benefit that could be accrued from transforming or severely degrading the land with unsustainable grazing or different kinds of cultivated crops. This amount is quite high and is based on the fact that farmers will maximise the use of the land, which is not necessarily the case. Although, farm gate prices were used, the costs will be relatively lower if production costs were deducted (although management costs would inflate the overall cost). However, a clearer picture of the forgone revenue of safeguarding biodiversity in the study area can be obtained by the difference between the benefits from sustainable and unsustainable use of the land (Balmford et al., 2002). This difference will be relatively smaller than the \$8.3 billion reported in this study, as sustainable use of the land can often generate considerable revenue. For example, meeting biodiversity targets for the study area also captures about 31 million tonnes of carbon and about 9700 LSU benefiting from fodder provision. At the time of this study, carbon was trading at prices ranging from US\$6.46 to US\$38.46 per tonne CO2 (http://www.ecosystemmarketplace.com, accessed June 2008). At a conservative price of US\$7.50 per tonne CO2 (about \$ 27 per tonne of carbon), the carbon captured in this study could produce an income of about a billion dollars (including transaction cost) for avoided carbon release (also see Naidoo and Ricketts, 2006). Fodder provision captured by the conservation plan would generate about US \$6.5 million. If revenue from other sources of income for sustainable land use is combined, the opportunity costs of conservation become smaller and land owners could receive financial benefit from sustainable use of land. However, management strategies aimed at producing high level of one ecosystem service might not necessarily produce high levels of the other.

Although there is the potential for land owners to benefit from sustainable use of the land, a shift in land use is only possible if the benefits are made clear. I have demonstrated three types of benefits. Carbon sequestration benefits are global, yet land owners can derive local benefits through international markets; water provision generates benefits for local people but no payment to land owners (unless some government or NGO decided to pay them); and sustainable grazing benefits the landowner directly. If the landowner gets money from a carbon sequestration project coupled with income from ecotourism and sustainable grazing, then the difference in cost of sustainable and unsustainable use will be narrowed and may not need intervention from a conservation agency. Naidoo and Rickets (2006) showed that conserving land could yield substantial benefit. In this study, I have shown that about \$1billion can be generated from carbon sequestration alone at a conservative value of \$ 7.50 per tonne CO2. This amount will be far greater if the upper value of \$38.46 per tonne CO2 is used. However, for sustainable use of the land for a public good like water recharge, the difference in the cost should be borne by the consumers or water conservation agency (e.g. department of water affairs and forestry).

What are the implications for implementation?

Conservation agencies and other stakeholders responsible for the management of ecosystem services such as water and grazing land can work together to safeguard ecosystem services and biodiversity in the study area where synergies exist. A carbon sequestration project is being tested in the Eastern Cape close to this study area. A call to donate money for this project to offset carbon emission from flying to the Society for Conservation Biology's 2007 Annual conference in South Africa was very successful (95% participation). This project gives hope that there may be potential for generating money from carbon-storage projects in the study area on top of other possible revenues from sustainable land uses such as ecotourism and sustainable grazing. However, where there is no synergy, a common ground for management can be found. According to Bohensky et al. (2004), making tradeoffs transparent to decision makers can assist the process of choosing between various options and the likely consequences of making alternative choices.

6.5 Conclusions

Systematic conservation planning for ecosystem services is still novel and refinement is needed of the tools and to concepts underpinning these initiatives. Other studies have reported limited or no congruence between biodiversity and ecosystem services when planned for separately (Chan et al., 2006), but I have shown the benefits of integrating the two objectives. I show that substantial amounts of ecosystem services can be captured in a biodiversity plan alone and even more can be conserved by including services into the plan or giving up very small percentages of our biodiversity targets for the study area for the same cost. This indicates that it is better to plan for biodiversity and ecosystem services simultaneously than planning for each separately and then looking for areas of synergy. However, planning for biodiversity and ecosystem services simultaneously will need different tools to those commonly used for biodiversity planning, such as MARXAN and C-plan with single reserve outcomes. A tool that can meet targets for ecosystem services and biodiversity simultaneously in different zones while seeking to maximise overlap will be most appropriate for planning for both. The commonly used MARXAN is being redesigned into MARXAN with zones to accommodate planning for multiple objectives. It is hoped that this tool will be useful in this regard.

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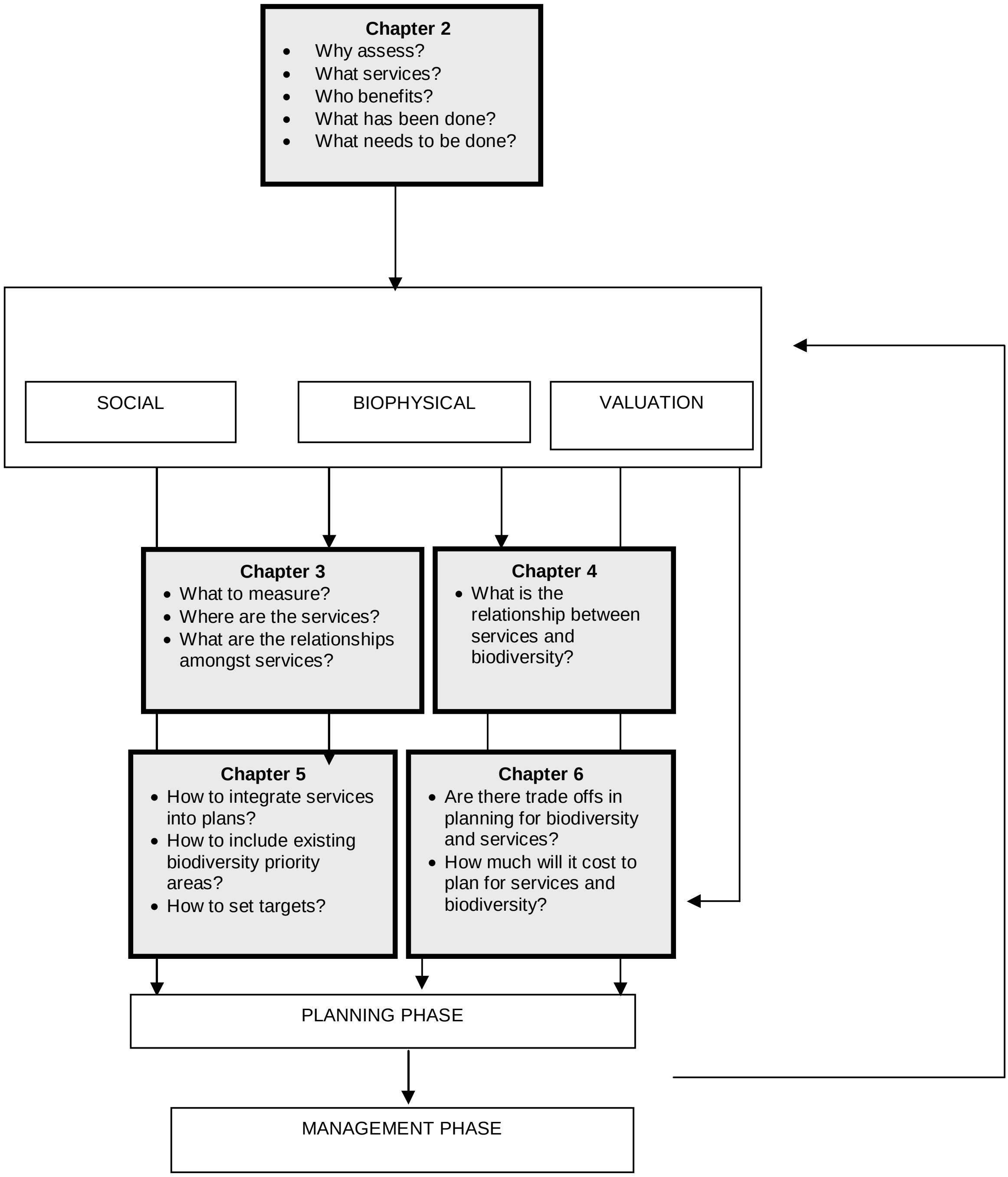
Chapter 2
Why assess?
What services?
Who benefits?
What has been done?
What needs to be done?
SOCIAL
BIOPHYSICAL
VALUATION
Chapter 3
What to measure?
Where are the services?
What are the relationships amongst services?
Chapter 4
What is the relationship between services and biodiversity?
Chapter 5
How to integrate services into plans?
How to include existing biodiversity priority areas?
How to set targets?
Chapter 6
Are there trade offs in planning for biodiversity and services?
How much will it cost to plan for services and biodiversity?
PLANNING PHASE
MANAGEMENT PHASE

CHAPTER 7. GENERAL DISCUSSION AND CONCLUSIONS

This thesis set out to investigate the topic of planning for ecosystem services. This chapter presents the main findings from the various studies comprising this thesis. I return to the main issues raised in chapter 1, and discuss the extent to which the work reported on here has contributed to the field. Where appropriate, priorities for further work are highlighted.

7.1 The extent to which ecosystem services are included in conservation assessments

Chapter 2 of this thesis showed that at present ecosystem services are not adequately captured in conservation assessments which are mostly geared towards the conservation of biodiversity in a narrower sense. While many studies use ecosystem services as a rationale for biodiversity conservation, only a few explicitly include ecosystem services. Those that include services consider only certain kinds of services such as water supply and carbon sequestration. It emerged in chapter 2 that supporting services (e.g. soil retention and formation) are often left out (only 13% of 16 included supporting services). More needs to be done to promote the inclusion of ecosystem services in plans, especially supporting services. The chapter also highlighted that although opportunities exist for an expansion of biodiversity conservation plans to also include ecosystem services, several potential constraints must be kept in mind to ensure the success of these plans. These include the fact that biodiversity and ecosystem services may not co-occur, are managed by different agencies and represent different value systems (intrinsic vs. utilitarian). With these opportunities and constraints in mind it was clear that an operational framework to inform planning for ecosystem services should be developed.

7.2 Developing methods for mapping ecosystem services

While there is interest in including ecosystem services in conservation plans in many regions, the lack of adequate data is a crucial handicap. The study in chapter 3 demonstrated that such data (or at least surrogate data) can be collated at a national scale. Biophysical data on ecosystem services can be developed using datasets often assembled for other purposes (e.g. water management, agricultural potential studies). In chapter 3, I showed that most data needed for mapping ecosystem services (precipitation, runoff, vegetation communities, species data, and other environmental variables) are readily available. The few studies that have mapped ecosystem services recently have made efficient use of such data sets (chapter 3; see also Chan et al., 2006; Naidoo et al., 2008). However knowledge of the services and how they work, together with robust mapping approaches and tools need to be developed to ensure that these data are used in a sensible manner. In chapter 3, a methodology for producing basic maps of ecosystem services was developed. Principles borrowed from conservation planning and biogeography proved very useful in developing these methods, relying on concept of ranges and hotspots to make the maps more meaningful. Maps of supporting services like soil retention and accumulation were produced for the first time in South Africa. Such contributions are critical to the development of more rigorous mapping procedure for services. These data are now available for use in subsequent plans for ecosystem services and for inclusion in biodiversity plans where necessary. The chapter made clear that a few key biophysical data sets, knowledge of the services under consideration, and concepts from conservation planning and biodiversity surrogates can ensure that data availability is not a limiting factor. It also made clear the need to engage with multiple disciplines (e.g. soil science, hydrology) in order to understand the services being mapped.

In this thesis, I have mapped only five ecosystem services from a list of 23 services as described by de Groote e al (2002). Other scientists have mapped a few more services not considered in this thesis such as outdoor recreation, aesthetic value, flood control, forage production, food production and pollination (van Jaarsveld et al, 2005; Chan et al., 2006). These services were mostly chosen due to the availability of data and it follows that the key to mapping ecosystem services is the capability of finding suitable proxies that can be used to map them. The supporting and regulatory services which have been mapped operate over intermediary spatial scales, are easier to define and proxies could be found to map them. The cultural services (e. g. spiritual and historic information, science and education) which operate from fine to broad scale (from one species to whole landscapes) may be more difficult to map unless they are clearly defined. More needs to be done in defining and finding proxies to map the remaining services as listed by de Groote et al. (2002).

7.3 Spatial congruence between ecosystem services

It became clear in chapter 3 that different ecosystem services are often provided by different areas, in a similar fashion to biodiversity where different areas are important to different taxa. This was further supported by the lack of congruence between services where no service can act as an ideal surrogate or proxy for the others. This finding corroborates those of Chan et al., (2006) who also found low levels of congruence between services in California. I showed that only the two soil services show some level of congruence (see Table 3.1). Primary productivity, although not a service per se, showed some potential as a surrogate for other services. The distribution of the five ecosystem services considered in chapter 3 also showed that most areas in South Africa produce some service, but few areas deliver multiple services. These findings call for an urgent assessment of congruence for a range of services, and for more study on the robustness of primary productivity (which is easy to map) as a surrogate for ecosystem services. Until this is done, no assumptions should be made regarding levels of congruence between services. The results from work presented in this thesis suggest that managing ecosystem services in a county like South Africa will be resource intensive. In other words, large areas will be needed to conserve the full range of important ecosystem services.

7.4 Spatial congruence between ecosystem services and biodiversity.

The debate on congruence between biodiversity and ecosystem functions has been going on for a long time (Tilman, 1996; Grime, 1997; Schwartz et al., 2000) and no clear answers have emerged. Some of these studies have reported a positive relationship between primary productivity and plant diversity whereby the most diverse communities are the most productive. However, few studies have examined the spatial dimensions of congruence between biodiversity and ecosystem services. The question on whether biodiversity and ecosystem services co-occur in space at difference levels of resolution is critical to safeguarding ecosystem services. This is one of the main questions that this thesis set out to answer.

The main findings in this thesis show that some ecosystem services (water flow regulation and supply) have a high overlap (about 70% in some cases) with biodiversity priority areas (see chapter 4). In chapter 5 a moderate overlap was also reported with biodiversity priority areas (fresh water and terrestrial) and ecosystem service priorities. These results support the findings of Tuner et al. (2007) who also found high overlap between biodiversity priorities and ecosystem services. Based on these results, it seems that biodiversity priority areas will capture some ecosystem services. However, whether this is the most efficient way to capture both biodiversity and ecosystem services, as well as the generality of these findings needs further clarification.

In addition to the congruence between overall biodiversity priority areas, I also set out to investigate the extent to which different biodiversity facets such as species and habitat are congruent with ecosystem services in chapter 4. Correlation between ecosystem service hotspots and those of species and habitat were generally positive but weak (mostly $r<0.2$, $p<0.05$). The strongest correlation was between soil accumulation and mammal hotspots ($r=0.27$, $p<0.05$). An interesting finding was the generally higher levels of mean species richness and vegetation diversity (at least 1.5 times higher than random in most cases) in the hotspot of water flow regulation in South Africa.

These overlaps although promising, are not necessarily an efficient way to conserve biodiversity and ecosystem services. So in chapter 6, I investigated the gains captured when ecosystem services and biodiversity were planned for simultaneously. Here a relatively high percentage of ecosystem services were captured (37-57%). These results suggest that win-win options for biodiversity and ecosystem services do exist in some instances, but one should definitely not conclude that the two are universally congruent. The general lack of congruence between biodiversity and ecosystem services as demonstrated in chapter 4, coupled with those reported by other studies (Chan et al., 2006; Turner et al., 2007; Naidoo et al., 2008) implies that conserving biodiversity will safeguard only certain ecosystem services. Clearly it is better to plan for ecosystem services and biodiversity simultaneously, as shown in chapters 5 and 6, than to rely on biodiversity surrogates as a conservation strategy for ecosystem services.

7.5 Integrating ecosystem services into a biodiversity plan

An important question in including ecosystem services is conservation planning is the cost implications of such a move. In chapter 6 of this study I showed that a great proportion of ecosystem services can be captured by a conservation plan based on biodiversity, by including data and targets for ecosystem services. By this, I mean, more ecosystem services could be captured at no additional cost to biodiversity by including data for ecosystem services.

Trading-off some of the biodiversity targets by a small amount also resulted in big gains for ecosystem services. Interestingly, I also showed that targets for ecosystem services could be increased without significantly increasing the cost of conservation. These results imply that ecosystem service conservation could benefit from biodiversity planning without necessarily increasing the cost to agencies. Not only could including ecosystem services in conservation plans be more cost efficient than planning for each separately, I also showed that when ecosystem services are included in a conservation plan in the Little Karoo study area, more than USD 1 billion could be generated from the storage of carbon captured by the plan. A further USD 6.5 Million could be generated from sustainable livestock production. Although not measured in the study in chapter 6, there are many other avenues to generate money from ecosystem services and sustainable land. This implies that some of the costs of conservation could be covered from money generated by the ecosystem services.

The opportunities for safeguarding ecosystem services and biodiversity through integrated conservation plans are enormous. First, conservation of ecosystem services can benefit from the current momentum for conserving biodiversity. Second, there is a general interest in considering ecosystem services in conservation assessments focussing on biodiversity. In chapter 2 I showed that many studies use ecosystem services as a rationale for biodiversity conservation. This is because of the benefit of using ecosystem services to generate funds while promoting biodiversity conservation efforts. Many feel that this is the best way forward for biodiversity conservation (Armsworth et al., 2007), but others think that using ecosystem services to promote biodiversity conservation might compromise the need to conserve biodiversity because of its intrinsic value (McCauley, 2006). Certainly, using ecosystem services to justify biodiversity conservation may backfire in areas where the value of biodiversity to humans is poorly defined. Despite the ongoing debate, conservation biologists are rightfully using every opportunity to secure biodiversity and to fulfil other environmental goals. An example is the payment for ecosystem services (PES) scheme, which is similar in structure to other incentive-based policies to achieve environmental goals (Pagiola, et al., 2005; Salzman, 2005; Wunder, 2005; Jack et al., 2008).

Payment for ecosystem services is an incentive-based conservation scheme where beneficiaries of ecosystem services are encouraged to pay providers of services (usually land owners who practice sustainable land use resulting in service provision). PES programs are widely used to encourage biodiversity conservation but also to promote sustainable use of land for the provision of ecosystem services. While these programs have great potential in environmental sustainability, they often require detailed information of which services are involved, where they come from, and where they are being used. In many cases, such information is not readily available. Maps of ecosystem services produced in this thesis (chapters 3 and 5) can be used to inform PES programs. There are opportunities to expand these programs and to improve scientific information that can feed into these programs and ensure continuous delivery of services and the persistence of biodiversity. In short, PES programmes and ecosystem service planning are mutually beneficial and should perhaps be more closely aligned

7.6 Identifying priorities for ecosystem services.

Although there may be benefits from including ecosystem services in conservation assessments geared towards biodiversity, priorities from such assessments may not necessarily conserve optimal level of ecosystem services. Safeguarding ecosystem services needs the identification of priority areas for services. The study in chapter 5 of this thesis makes a valuable contribution in this regard. Priorities were identified for different services in the grassland biome of South Africa. An important contribution is the identification of priorities for a set of services and the integration of biodiversity in such priorities in the study area. The results showed that tools used in planning for biodiversity such as MARXAN can be used in planning for ecosystem services. However, challenges around determining targets for ecosystem services remain significant and are an area of much needed research and agreement. The integrated plan for biodiversity and ecosystem services showed that it was possible to meet targets for ecosystem services in areas that align with biodiversity priorities.

Planning for ecosystem services is an emerging field and there are many challenges. Although this thesis has shed light on some issues, there is still a way to go. First, methods for mapping and quantifying services need to be standardized. For example, the few studies that have mapped the services of water provision have used different proxies. Second, ecosystem services are should not be included into conservation planning is an ad-hoc manner. They need careful considerations about targets that will meet human needs and ensure the continuity of service delivery. Third, the multidisciplinary nature of the quantification of ecosystem services and the lack of congruence amongst services and with biodiversity implies that managing ecosystem services will be demanding in terms of resources and knowledge. Fourth, even when good scientific information becomes available, converting science into action on the ground will need much effort. Appropriate policies must be put in place that work for ecosystem services but that are not detrimental to biodiversity. Nelson et al. (2008) showed that policies aimed at increasing the provision of ecosystem services can, but do not necessarily, increase the provision of biodiversity (in this case species) conservation. They note that considering a range of ecosystem services in a conservation policy will only magnify the degree of potential tradeoffs. Lastly, the mainstreaming of ecosystem services for implementation as proposed by Cowling et al. (2008) is crucial, but will require time to be incorporated into existing and newly developing policy frameworks.

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Appendix A :
An operational model for mainstreaming ecosystem services for implementation

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ABSTRACT

Research on ecosystem services has grown markedly in recent years. However, few studies are embedded in a social process designed to ensure effective management of ecosystem services. Most research has focused only on biophysical and valuation assessments of putative services. As a mission-oriented discipline, ecosystem service research should be user-inspired and user useful, which will require that researchers respond to stakeholder needs from the outset and collaborate with them in strategy development and implementation. Here we provide a pragmatic operational model for achieving the safeguarding of ecosystem services. The model comprises three phases: assessment, planning, and management. Outcomes of social, biophysical, and valuation assessments are used to identify opportunities and constraints for implementation. The latter then are transformed into user-friendly products to identify, with stakeholders, strategic objectives for implementation (the planning phase). The management phase undertakes and coordinates actions that achieve the protection of ecosystem services and ensure the flow of these services to beneficiaries. This outcome is achieved via mainstreaming, or incorporating the safeguarding of ecosystem services into the policies and practices of sectors that deal with land- and water-use planning. Management needs to be adaptive and should be institutionalized in a suite of learning organizations that are representative of the sectors that are concerned with decision-making and planning. By following the phases of our operational model, projects for safeguarding ecosystem services are likely to empower stakeholders to implement effective on-the-ground management that will achieve resilience of the corresponding social-ecological systems.

Keywords: adaptive management, land-use planning, social ecological systems, stakeholder engagement.

Introduction

There has been an impressive growth in research on ecosystem services in recent years. However, few studies are embedded in a social process designed to ensure effective on-the-ground management of areas that deliver ecosystem services. It is unlikely that the outcomes of technically sophisticated assessments published in scientific journals will lead to implementation via a ‘‘trickle-down’’ effect (1–3). As a mission-oriented, pragmatic discipline (4), ecosystem service research should be geared for implementation, and scientists should assist this process by responding to stakeholder needs from the outset and by becoming involved in the messy process of collaborating with and empowering stakeholders in strategy development and implementation (1, 5–7). How to do this is the topic of this article. There are some excellent examples of research that have resulted in the protection of ecosystem services (e.g., refs. 8–10). But they are few and are cited repeatedly in the literature. Our wish is that ecosystem service research does not become another bandwagon driven by technological sophistication and characterized by societal irrelevance. As a cornerstone of sustainability science (11), ecosystem service research needs to be user inspired, user-useful, and user-friendly. Although research-for implementation models exist for integrated natural resource management (7) and conservation planning (5), we know of no article that spells out pragmatically and comprehensively the process for achieving the safeguarding of ecosystem services on the ground. Our article seeks to fill this gap. To provide a real-world context, we have chosen to focus on the internalization, or ‘‘mainstreaming’’ (12), of ecosystem service concerns into the land-use (and water-use) planning sector. Land-use planning is a normative discipline (4) in the sense that it provides the legally entrenched norms and rules for making decisions about how natural resources are to be used. In many parts of the world, governments are institutionally obliged to iteratively conduct participatory, spatially explicit, land-use planning aimed at integrating requirements for social, economic, and environmental sustainability. Flaws notwithstanding (13), this process provides a window of opportunity for mainstreaming ecosystem services into the activities of organizations that are empowered to make routine decisions about the use of land and water resources (14, 15). We restrict ourselves to ecosystem services—defined as the end products of nature that benefit humans (16)—provided by natural and semi-natural habitats (wild nature). Thus, we do not consider agriculture or aquaculture ecosystems, acknowledging, of course, that wild nature does provide services essential for the success of these ecosystems. First, this article provides some background on mainstreaming, a relative newcomer to the biodiversity lexicon. The second and substantive part provides a pragmatic, operational model for guiding the things we need to do for implementing the safeguarding of ecosystem services. Our account draws on our collective experience over the past decade in user-inspired research and implementation in the nature conservation and water sectors (e.g., refs. 14 and 17–20).

What Do We Mean by Mainstreaming?

In the context of natural resource management and conservation, the objective of mainstreaming is to internalize the goals for safeguarding resources into economic sectors and development models, policies, and programs, and therefore into all human behavior (12). The concept is entrenched in several articles of the Convention on Biological Diversity and is the explicit objective of the Global Environmental Facility’s GEF-4 program, with its particular emphasis on ecosystem services.

Based on South African experience, there are four elements of a framework for achieving: mainstreaming: (*i*) prerequisites, elements without which mainstreaming cannot happen; (*ii*) stimuli (or windows of opportunity), elements external and internal to the sector that catalyze awareness of the need for mainstreaming; (*iii*) mechanisms, the actual activities that seek to effect mainstreaming; and (*iv*) outcomes, the measurable indicators of mainstreaming effectiveness (20). The most frequently cited prerequisites in these projects were democratic and accountable governance, awareness and knowledge, and organizational and institutional capacity. Mainstreaming is achieved primarily through behaviour change. In the context of this article, it requires that the safeguarding of ecosystem services is institutionalized in landuse planning policies and is reflected in the day-to-day activities of this sector.

Operational Model

Here we discuss operational issues: the things that need to be done to mainstream ecosystem services. Fig. A.1 shows the three phases of the model (assessment, planning, and management) and their relationships to spatial scale, degree of stakeholder engagement, and status of the social-ecological system—the integrated and interactive relationships between humans and ecosystem services (21, 22). Our operational model is based on one devised by Knight *et al.* (5) for conservation planning. Any project for safeguarding ecosystem services should strive to arrive at the top right-hand corner in Fig. A.1, where the adaptive management of the social-ecological systems associated with the defined suite of ecosystem services has been mainstreamed into an appropriate land-use planning framework and governed by learning organizations that are representative of, and supported by, the full range of stakeholders in the study area (a learning organization is one skilled at creating and acquiring knowledge and modifying its behavior to reflect new insights) (23). Thus, stakeholders are empowered to implement effective on-the ground management of ecosystem services, and social-ecological systems are resilient (they can absorb shocks and surprises) (1, 22, 24). Getting there is a social process riddled with complexity, contention, uncertainty, surprise, disappointment, and triumph. It will take a long time—in many cases, decades—to achieve this goal (25). Below we describe some elements of this pathway to resilience by outlining the key actions associated with each of its major phases: assessment, planning, and management.

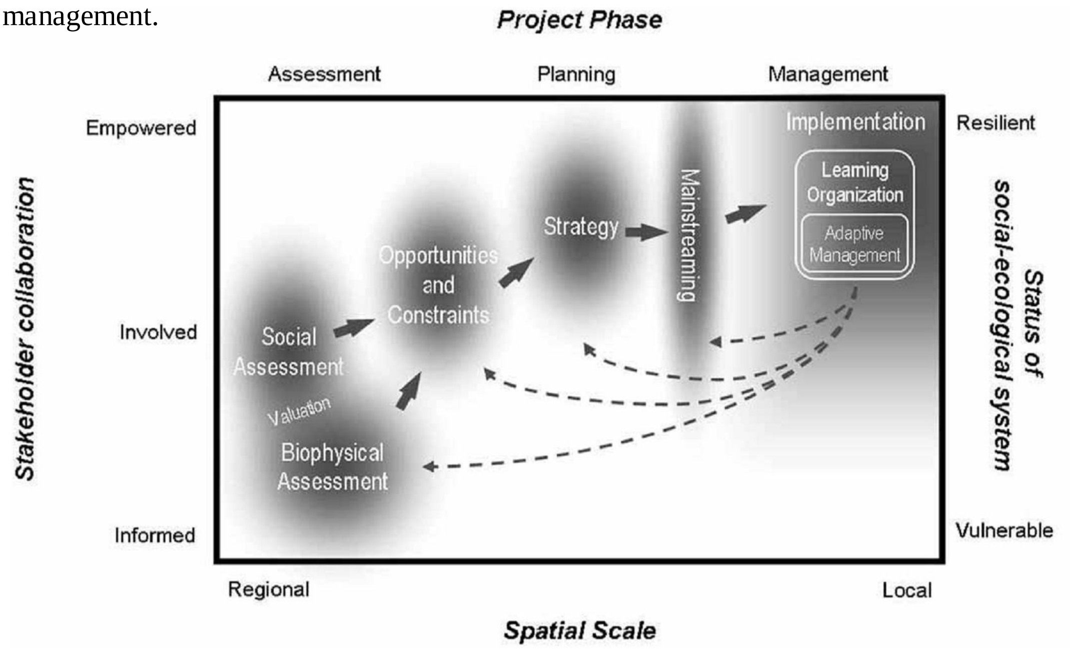

Figure A.1. An operational model for implementing the safeguarding of ecosystem services.

Assessment Phase. The assessment is a structured process that provides knowledge useful for policies, strategies, and management but does not prescribe these (5, 14, 26). The assessment seeks to answer questions inspired by the beneficiaries and managers of ecosystem services; in our situation, it must provide knowledge useful for mainstreaming ecosystem services into local land-use planning. We identify three types of assessment: social, biophysical, and valuation (Fig. A.1). A key requirement of the assessment phase is the establishment of multidisciplinary and multisector teams (5, 27, 28). Although teams engage in empirical research, their activities should be coordinated by the goals defined for ecosystem services research, which, in turn, are defined by the requirements of land-use planning, a normative discipline (4). This hierarchy of coordination provides operational meaning to the notion of ecosystem service research as a truly interdisciplinary activity (4).

Teams should include researchers from the natural and social sciences and the humanities; scientists and managers from the natural resource management (water, fisheries, agriculture, forestry, conservation, etc.) and human well being (health, social development, safety and security, land-use planning, etc.) sectors; and nongovernmental and other citizen-based organizations. In addition to data collected by using standard scientific methods, assessment teams also should record tacit (or implicit) and traditional knowledge because a great deal of useful information is associated with these informal systems (29, 30). Teamwork is both difficult and rewarding. It requires emotionally intelligent leadership, which is rare. There may be confusion and contention about values assigned to nature: conservationists typically view nature as axiomatically ‘‘good,’’ whereas other stakeholders perceive the value of nature in a relative sense (31). This kind of confusion needs to be managed by effective leaders, as do power asymmetries and concealed agendas. However, if properly managed, teamwork provides excellent opportunities for rapid, collaborative learning based on both explicit and tacit knowledge and for challenging or changing deeply entrenched world views or mental models (3, 30).

Social assessment. The social assessment should precede the biophysical one (Fig. A.1) because it identifies the owners and beneficiaries of ecological functions that actually deliver services and, hence, require biophysical assessments. It also identifies markets for ecosystem services and other incentives for their safeguarding, as well as individual, institutional, and governance barriers to implementation (32). The social assessment should provide knowledge on the needs, values, norms, and behaviors of individuals, institutions, and organizations in the study area. In other words, it provides an understanding of how an area works in socioeconomic terms and why. Without the understanding of the social system provided by the social assessment, implementation is likely to be poorly targeted. Specific issues requiring research will vary with context; however, knowledge of the spatial patterns of population density, human needs (for example, subsistence, protection, and identity), income distribution, current and future trends in land use, land prices, infrastructure, the social capital of natural resource management organizations, nature-related values, preferences and ethics, and incentives for behavior change are likely to emerge as important topics in most cases (26, 33–36). Wherever feasible, data need to be captured spatially and matched to the scale used in the biophysical assessment (37). Social assessments take time and can be costly. Adequate budgets should be secured (38, 39).

Biophysical assessment. Biophysical assessment provides knowledge on the types and location of the biophysical features that provide ecosystem services, the spatial and temporal flows of services in relation to beneficiaries, and the impacts of land and water transformation on delivery (e.g., refs. 40–44). Heal (33) makes the important point that it is the biophysical rather than the valuation assessment that provide the knowledge-based case for safeguarding services. For example, a simple model that predicts the reduction of water supply below the projected demand as a consequence of unchecked devastation by humans of a watershed (e.g., refs. 8 and 9) is likely to provide a more compelling case to stakeholders for protecting the watershed than dubious estimates of the reduction in the aggregated monetary value of all of the watershed's goods and services (most of which have no market value). Other than research on the links between biodiversity and ecosystem services (45), very little research has been done on the ecology of ecosystem services. Kremen (40) provides a useful operational framework for studying ecological aspects of ecosystem services that we do not repeat here other than to reiterate—as others have done (e.g., refs. 43 and 46)—the importance of measuring the spatiotemporal scales over which services operate. Of the published biophysical assessments, most focus on mapping services, their flows, and the impacts of habitat transformation on these flows (e.g., 41–44). A feature of many studies is the identification and mapping of natural features that have no direct beneficiaries or markets and in whose protection few people have an obvious interest. Invariably, these studies are not user-inspired and lack social assessments for identifying the suite of services that fulfils social needs, both presently and potentially. In short, without beneficiaries, there are no services. An important component of the biophysical assessment is the development of dynamic models of landscape change—the spatially explicit depiction of alternative futures (38, 47). These products allow stakeholders to envision the consequences of particular policy frameworks regarding land and water use. However, they need to be interpreted visually and depicted as plausible scenarios that stakeholders comprehend. We return to this very important point later on in this article.

Valuation assessment. Just as the study of the nature-related values (beliefs) that people hold is contentious, so is the study of the values they assign (in monetary or ranking terms) to nature (48). There is a large and growing literature on the conceptual, technical, and operational aspects of the economic valuation of ecosystem services (e.g., refs. 49–51). We do not review typologies and techniques for economic valuation but rather focus on several issues that are relevant for a socially engaged valuation assessment. The valuation assessment is located at the intersection of the social and biophysical assessments (Fig. A.1) and should be informed by these (46). Most studies advocate, albeit with caution, the monetary valuation of ecosystem services because ‘‘most societies have an intuitive notion of economic value’’ (51) and it provides ‘‘a metric than can be deployed across competing land uses’’ (52). Monetary valuation can be particularly effective in enabling informed tradeoffs in cost–benefit analyses, where the focus is on assessing the marginal change in the provision of an ecosystem service that has market value (e.g., amount of water produced) relative to a competing land use that also is traded on the market (e.g., real estate) (33, 52). But the vast majority of services have no market price (33, 41, 46, 52). To paraphrase Simpson (49), prices are not to be confused with values, and prices are not the only values that are important. Nonmonetary units of value also can be used, for example, cubic meters of clean water, jobs created, and lives saved (27). Because money is the most commonly used interchangeable commodity, valuation in monetary terms may send the message that a service is more easily replaced by human manufactured providers than it actually is (53). Throughout the world, land-use decisions are seldom made on the basis of the outcomes of economic valuation studies; they usually are made by officials and politicians—many of whom are poorly informed—or, in functional democracies, by variously informed citizens. We recommend, where circumstances permit, encouraging stakeholders to reach consensus on assigning subjective values to ecosystem services. Such discourse-based approaches (54, 55) enable social influence and consensus to define knowledge about the value of ecosystem services. As Starbuck (3) states: ‘‘Acceptance by people is crucial, because knowledge is what people say it is.’’

Identify opportunities and constraints for implementation.
The concluding stage of the assessment phase is a structured process in which all project participants identify opportunities and constraints for implementing actions to safeguard ecosystem services. Because of different value systems, research traditions, and mental models, this process can be difficult (38, 39, 56); it requires excellent facilitation and leadership. We cannot overstate the importance of this phase: the outcomes provide the bridge between assessment and planning by providing knowledge essential for identifying strategic implementation objectives (Fig. A.1). Identifying opportunities and constraints can be challenging because of the complex outcomes of the three assessments. There is a need to

frame and depict in ways that harmonize with stakeholders' values, needs, and cognitive skills, the complexity of outcomes characterized by situations of ''numerous possible futures underpinned by numerous possible solutions'' (57). Most stakeholders are likely to lack the cognitive capacity to comprehend and absorb the significance of models that depict dynamic, long-term, continuous, and multiscale processes with complex feedback and uncertain outcomes (3, 29, 58). They find it much easier to relate to models that are described by discrete events, possibilities, pictures, emotions, and stories, and that provide prospects for harnessing their energies and skills (59–62).

Scenario planning is one way that the assessment team can display implementation opportunities and constraints in a manner that is comprehensible to a broad range of stakeholders. This powerful tool deals with uncertainty by providing plausible, descriptive narratives or pathways to the future. Scenario planning has a long history in business science where it has been used to challenge mental models, facilitate behavior change, promote collaborative learning, and confront tradeoffs (56). It also has been used to good effect in the natural resource sector (7, 63, 64) and was adopted by the Millennium Assessment (65, 66). In the context of our model, scenario assumptions are defined by implementation opportunities and constraints, and these are used to set parameters for spatially explicit models of alternative futures that can be depicted as maps and visual narratives (38). Scenarios can be especially effective when they capture alternative futures visually and dramatically, in such a way as to reduce stakeholder confusion by providing clarity about complex issues and vague language (67, 68). By providing compelling, positive alternatives to the status quo, scenarios can harness stakeholders' energies for strategy development and, thereby, overcome their sense of helplessness about the future (69–72).

Planning Phase. The second phase of the operational model is planning, which is explicitly collaborative, involving all key stakeholders, including researchers (Fig. A.1). Collaborative planning is a discourse-based process that comprises the identification of a vision, a strategy to realize this vision, specific strategic objectives, and instruments, tools, and organizations for implementing actions to achieve the objectives.

Strategy development. The overall aim of this stage of the planning process is to collaboratively identify a set of strategic objectives and specific actions for the safeguarding of ecosystem services. These objectives should seek to exploit the implementation opportunities and overcome the constraints identified in the assessment phase. Scientists need to develop and present at the strategy workshops products (for example, visually compelling scenarios and maps) that are user-useful and user-friendly (5, 14). Strategy development is essentially a process for learning (56, 73)—an opportunity for non experts to gain an understanding of the issues at stake and for experts to appreciate the concerns and contributions of other stakeholders, including decision makers and the socially marginalized. The involvement of non experts also is an important opportunity to engender pro-nature behaviour change: appropriately framed information and involvement in a process of developing a strategy to achieve a mutually desired state—the vision—can rapidly change people's norms (62, 74, 75). It forces them to confront realities about unsustainable futures that will be harmful to themselves and their offspring and to contribute by exploring possible solutions to these problems (72, 75–77). In the strategy process, scientists are enablers (5, 26). They need to frame issues clearly and communicate in simple and accessible language the benefits and costs of particular actions and their associated uncertainties (58, 67, 78–80). Their role is to help stakeholders understand issues so as to avoid confusion and overcome helplessness (69, 71). The strategic objectives need to be an unambiguous and tractable list of actions and behaviors that are clearly linked to instruments for implementation, which are supported by appropriate institutions (5, 81). The instruments available will be context-specific and, because many instruments are complementary, they should be identified as an optimal mix (5). They may include financial incentives (e.g., direct and indirect payments for service delivery), governance-based instruments (e.g., enforcement of existing legislation, capacity-building, and the establishment of cooperative governance structures), and value based instruments (education and recognition) (28, 29, 38, 52, 82, 83). In the cases where markets exist for ecosystem services—for example, carbon sequestration, nature-based tourism, and water supply—institutions and organizations may need to be established to capture the values of these (52, 83).

Mainstreaming. Mainstreaming, the internalization of ecosystem service safeguarding into the policies and practices of the land-use planning sector, is located at the interface of the planning and management phases of the operational model (Fig. A.1). Optimal mainstreaming requires effective governance, organizational and institutional capacity, and awareness of and a comprehensive knowledge about the ecology and value of ecosystem services (20). The assessment and planning phases provide knowledge about ecosystem services, increased awareness of the importance of these services among stakeholders (and may have already initiated a change in mental models or even behavior), and identify opportunities and constraints regarding governance and capacity for implementation. The rationale, benefits, and mechanisms for safeguarding ecosystem services need to be mainstreamed into all of those sectors that feed into land-use planning, e.g., water, forestry, agriculture, tourism, and urban planning. At least three things need to be considered when launching a mainstreaming initiative. First, decision makers in all of the relevant sectors need to be made aware of the importance for sustaining society of safeguarding ecosystem services and, where they exist, of their legal mandates to do so, which is most effectively done by identifying ''win–win'' situations that address both natural resource and socioeconomic concerns (12). For example, this was done to mainstream restoration projects in South Africa that delivered on both ecosystem service and social equity goals (84, 85). Communication to decision makers must be effective (78, 80); it often may be necessary to emphasize as compelling ‘‘sound bites’’ the immediate, social, and economic benefits of ecosystem service protection (59) rather than less certain benefits that may only manifest in the longer term. Second, new organizations and institutions will be required to address the tricky problem of coordinating governance across such a wide array of sectors (22). We discuss cooperative governance in the next section of this article. Third, increased awareness and knowledge about environmental concerns, and even embracing pro-nature values, does not necessarily translate into adopting pro-nature behavior (62, 86). Therefore, pragmatic solutions are required to overcome the inertia in engendering pro-nature behaviors of individuals and organizations that are required for mainstreaming. Social marketing is very promising in this respect: rather than attempting to understand the complex causes of behavior, it takes existing behaviors as a given and then seeks to identify the barriers to behavior change and to design specific incentive based programs to overcome these barriers (86, 87). Incentives relate to both internal barriers (e.g., absence of skills, opposing values, and beliefs) and external barriers (e.g., inadequate infrastructure and support). Social marketing has been extremely successful in achieving behavior change in the health, social development, and waste management sectors but has yet to penetrate natural resource management and conservation sectors. Depending on the outcome of the assessment of governance and institutional capacities, it may be necessary to implement programs of social marketing to bring about rapidly the desired levels of behavior change. Mainstreaming is an ongoing process that needs to be responsive to windows of opportunity and other unintended surprises arising from, among others, market emergence, infrastructure development, and political changes and associated shifts in power regimes (20, 26).

Management Phase. Management comprises the final phase of our operational model for achieving resilience of the social ecological systems associated with ecosystem services. The overall objective of this phase is to undertake and coordinate actions, including additional research, that achieve the protection of biophysical features that provide ecosystem services and ensures the flow of services to beneficiaries. Actions may include the implementation of social marketing projects, the restoration of vegetation for carbon credits, the protection of watersheds key for water delivery, or the protection of viewsheds for nature-based tourism—it depends on what has emerged from the assessment of implementation opportunities and constraints. We recommend, as others have done (1, 14), the adoption of an adaptive management framework that embodies an action-reflection cycle, or ‘‘learning by doing’’ (24, 30). In this regard, the adoption of a quasi-experimental approach, whereby the effectiveness of interventions can be assessed relative to situations where intervention is withheld (88), can be extremely effective in unravelling the complexities of social-ecological systems (14). Static products such as user-useful and user-friendly maps of ecosystem services and guidelines for managing them, which can be mainstreamed directly or via social marketing into local integrative planning processes, potentially are very useful (e.g., 14). Adaptive management needs to respond effectively to the complex feedback, opportunities, and shocks that characterize social-ecological systems and provide insights that can be incorporated into the iterative processes of assessment and planning (Fig. A.1). Therefore, adaptive management needs to be institutionalized in a suite of learning organizations (5, 14, 28, 78), each focusing on a different ecosystem service. Such organizations must be representative of the sectors that are concerned with land-use decision-making and planning and should foster a spirit of co-learning, co-governance, and accountability (22, 23, 56), which is not always easy to achieve (19); key individuals and good leadership are of paramount importance for effective learning organizations (22, 89). The learning organization should have the authority to restrict access to ecosystem service providers, the wherewithal to offer incentives for their safeguarding, the capacity to monitor ecological and social conditions, the expertise to evaluate the outcomes of interventions, and sufficient flexibility to respond rapidly to changed circumstances (35, 57).

Conclusions and Caveats

At the core of our operational model are three elements: socially relevant, user-inspired research, stakeholder empowerment, and adaptive management embedded in learning organizations. The goal is the achievement of social and ecological resilience in an uncertain world. The activities prescribed by the model will not be easy to implement. Socially engaged, multi- and interdisciplinary research is relatively rare. Our process requires a fundamental change, or transformation, in the way research generates knowledge (3, 4). Researchers will need to be responsive to stakeholder needs, collaborate with many groups with values and norms foreign to their own, operate as facilitators of knowledge transfer to stakeholders, and be prepared to engage time consuming processes that are not sympathetic to career aspirations and performance benchmarks predicated by the accumulation of publications in high-impact journals (7, 90). Moreover, the education philosophies of almost all universities are not conducive to multi- and interdisciplinary research; instead, they encourage the atomization of disciplines and entrench the boundaries between them (4, 58, 91). However, the recent emergence of sustainability science (11) is a very positive development. The operational model presented here provides many opportunities for conducting research on the complex problems inherent in managing social-ecological systems. Recognition of the importance of this research through enhanced funding and status can provide the impetus for its growth. Implementing the operational model for most projects will take a lot of time (25) and incur large costs, especially transaction costs (92, 93). In developing countries, donor organizations fund projects that are geared to specific deliverables subject to the time-related tyrannies of log frames, which may not be appropriate for ecosystem service projects. Our operational model is a process that does have hallmarks for evaluation but is simply too complex and uncertain of outcomes to specify, with any degree of realism, tangible outputs in short (1- to 5-year) timeframes. The operational model specifies a process that engenders stakeholder collaboration and bottom-up decision-making, which is consistent with the notion that although most environmental problems are regional or global, the solutions are at the local and individual scales (94). However, there are many cases where well intentioned, bottom-up projects fail because of failures of regional and global institutions to support their outcomes (25, 95, 96). Bottom-up implementation needs to be complemented by the policies and practices of regional and global trade and financial institutions (97). Of great importance is the incorporation of the value of ecosystem services into the accounting systems of these institutions (16). Related to this is the need to project ecosystem services research into the realm of transdisciplinarity by addressing directly the values, ethics, and morals associated with individuals, organizations, and institutions (4, 92, 98). Do we want a world that promotes wealth accumulation and self-interest, or one that fosters equity and common good? Questions such as these raise issues about the kinds of economic systems we desire: ones based on perpetual growth or ones that strive for a steady state (98, 99). Sadly, the prevailing consumerist economic paradigm, the high discount rates held by most humans, and their disconnect for the natural world (58, 69, 100) do not augur well for the radical transformations required to place the world on a path to sustainability. Planning to ensure the persistence of ecosystem services in guaranteed to be an important and stimulating challenge.

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