ASSESSING BIODIVERSITY INTACTNESS

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Assessing biodiversity intactness

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Declaration

I declare that this dissertation is my own work, except where acknowledged. It is being submitted for the Degree of Master of Science to the University of the Witwatersrand, Johannesburg, South Africa. It has not been submitted before for any degree or examination in any other university.

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Abstract

Biodiversity, currently being lost at rates far in excess of mean rates estimated from the fossil record, is key to the functioning of ecosystems and the provision of ecosystem services, and thus essential to human well-being. Consequently, the nations of the world have set themselves a target of reducing the rate of biodiversity loss by 2010. However, no currently used measures of biodiversity are adequate for assessing progress towards this target. This study proposes an index that is simple and practical, yet sensitive to important factors that influence biodiversity status, and which satisfies the criteria for policy relevance set by the Convention on Biological Diversity. The Biodiversity Intactness Index (BII) is an indicator of the average abundance of all species of the well-known terrestrial biodiversity (plants, mammals, birds, reptiles and amphibia) relative to a reference level. For large parts of the world, including southern Africa, populations prior to the alteration of the landscape by modern industrial society serve as a meaningful reference. The BII integrates across the differential impact of a range of land use activities, taking account of the species richness of the areas affected, to give an average impact for a given area and time.

Application of the BII on a 4-million km^2 region of southern Africa gives a score of 84 ± 7% for the year 2000. In other words, averaged across all plant and vertebrate species in the region, populations are estimated to have declined by 16% relative to their pre-colonial levels. National scores range from 69% for Lesotho to 91% for Namibia. The taxonomic group with the greatest loss is mammals at 71% remaining, and the ecosystem type with the greatest loss is grassland at 74% remaining. The rate of loss of BII for the decade of the 1990s is estimated as 0.8% absolute. A scenario analysis for the 21st century suggests that the rate of decline is unlikely to decrease significantly until the latter third of the century, with BII scores ranging from 56% to 68% by the year 2100. In South Africa, the BII has declined from 91% to 80% over the 20th century, and the relationship between BII and Gross Domestic Product (GDP) suggests that economic development has become progressively decoupled from biodiversity loss. These results suggest that targets for reducing the rate of biodiversity loss need to take account of the stage of development in different regions.

List of Publications

All chapters in this dissertation, other than the introductory and concluding chapters, are in the form of stand-alone scientific publications. The versions that appear in this dissertation are current at the time of submission. For authoritative and citation purposes, please refer to the final published versions of the chapters:

Chapter 2: Biggs, R., Scholes, R.J., ten Brink, B. and Vačkář, D. (*In press*). Biodiversity Indicators. In: *Sustainable Development: How to measure progress through indicators* (Moldan, B., Hak, T. and Bourdeau, P., eds.). Island Press: Washington, D.C.

Chapter 3: Scholes, R.J. and Biggs, R. (2005). A biodiversity intactness index. *Nature* **434**, 45-49.

Chapter 4: Biggs, R., Scholes, R.J. and Reyers, B. (2004). Assessing biodiversity intactness at multiple scales. Paper presented at the Bridging Scales and Epistemologies Conference 17-20 March 2004, Alexandria, Egypt[1].

Chapter 5: Biggs, R. Reyers, B. and Scholes, R.J. (*Submitted*). A biodiversity score for South Africa. *South African Journal of Science.*

Chapter 6: Simons, H., Biggs, R., Bakkenes, M., Scholes, R.J., Eickhout, B., and van Vuuren, D. (*In prep.*). Biodiversity changes in southern Africa in the 21st century. For submission to *Ambio*.

Chapter 7: Biggs, R. and Scholes, R.J. (*In press*). Historical changes in natural capital in South Africa: An approach using changes in biodiversity. In: *Restoring Natural Capital: Views from the South* (Milton, S.J. and Aronson, J., eds). Island Press: Washington, D.C.

Publications resulting from the Millennium Ecosystem Assessment

The work carried out in this dissertation forms part of the Southern African Millennium Ecosystem Assessment (SAfMA), which is in turn linked to the global Millennium Ecosystem Assessment (MA) initiative. The MA and SAfMA publications to which I have contributed, and in which aspects of this work are reported on, are:

[1] www.millenniumassessment.org/en/about.meetings.bridging.proceedings.aspx

- Capistrano, D. Lebel, L., Samper, C., Scholes, R.J., Wilbanks, T.J., Biggs, R., Lee, M.J., Petschel-Held, G. (2003). Dealing with scale. In: *Ecosystems and Human Well-being: A Framework for Assessment* (Millennium Ecosystem Assessment). Island Press, Washington DC.
- Zermoglio, M.F., van Jaarsveld, A.S., Reid, W.V., Romm, J., Biggs, R., Yue, T.X. and Vicente, L. (*In press*). The multiscale approach. In*: Sub-global assessments* (Millennium Ecosystem Assessment). Island Press, Washington D.C., USA.
- Lebel, L., Thongbai, P., Kok, K., Agard, J., Bennett, E., Biggs, R., Ferreira, M., Filer, C., Gokhale, Y., Mala, W., Rumsey, C., Velarde, S.J., Zurek, M., Blanco, H., Lynam, T. and Yue, T.X.. (*In press*). Sub-global scenarios. In*: Sub-global assessments* (Millennium Ecosystem Assessment). Island Press, Washington D.C., USA.
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- Biggs, R., Bohensky, E., Desanker, P. V., Fabricius, C., Lynam, T., Misselhorn, A. A., Musvoto, C., Mutale, M., Reyers, B., Scholes, R. J., Shikongo, S. and van Jaarsveld, A. S. (2004). *Nature supporting people: The Southern African Millennium Ecosystem Assessment.* Pretoria, South Africa: Council for Scientific and Industrial Research.
- Van Jaarsveld, A.S., Biggs, R., Scholes, R.J., Bohensky, E., Reyers, B., Lynam, T., Msuvuto, C. and Fabricius, C. (*In press*). Measuring conditions and trends in ecosystem services at multiple scales: The Southern African Millennium Assessment (SAfMA) experience. *Royal Society proceedings: Beyond extinction rates: Monitoring wild nature for the 2010 target.*

Other publications

Aspects of the work presented in this dissertation, or on which this dissertation draws, also appear in the following publications to which I have contributed:

- Biggs, R. and Scholes, R.J. (2002). Land-cover changes in South Africa 1911-1993. *South African Journal of Science* 98(9/10): 420-424.
- Biggs, R. and Scholes, R.J. (*In press*). Southern Africa. In: *The Earth's Changing Land: An Encyclopedia of Land-Use and Land-Cover Change* (Geist, H. ed.), Greenwood Publishing Group, Westport, USA.
- Biggs, R. and R.J. Scholes. (In press). South Africa. In: *The Earth's Changing Land: An Encyclopedia of Land-Use and Land-Cover Change* (Geist, H. ed.), Greenwood Publishing Group, Westport, USA.
- Scholes, R.J., W.K. Küper and R. Biggs. (*In press*). Biodiversity. In: *Africa Environmental Outlook 2* (UNEP ed.), Nairobi, Kenya: UnitedNations Environment Programme.

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Chapter 1: Introduction

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As global environmental change assumes a more central place in human affairs, science is being thrust into the unfamiliar and uncomfortable role of a primary participant in a heated and potentially divisive international debate about the nature and severity of global change and its implications for ways of life. Much is at stake, and the game is being played hard. Despite the risks, science must accept the responsibility of developing and communicating the essential knowledge base societies can use to consider and decide ultimately on how to respond to global change.

Berrien Moore III (2002)

1.1 Global environmental change and biodiversity

Human activities are now so pervasive, and their impact on the Earth System so profound, that they threaten the survival of many species, and the well-being of humans themselves. In terms of several key environmental parameters, the Earth System has moved well beyond the range of natural variability exhibited over at least the past half million years. The nature, magnitude, and rates of the changes currently taking place are unprecedented. They affect all components of the Earth System: the land; coastal and marine environments; the atmosphere; the water, carbon, nitrogen, phosphorous and other elemental cycles; and the Earth's biological diversity. These linked changes are collectively referred to as *global change* (Walker *et al.* 1999; Steffen *et al.* 2004). For example, the vegetative cover of nearly half the Earth's land surface has been transformed by direct human action, with significant consequences for biodiversity, nutrient cycling, soil structure and biology, and climate (Turner *et al.* 1990). More than half of all accessible freshwater on Earth is appropriated for human purposes (Postel *et al.* 1996). Extinction rates are increasing sharply in marine and terrestrial ecosystems around the world, with the present 'sixth great extinction' event in the Earth's history being the first caused by the activities of a biological species, and not by forces of nature or extraterrestrial sources (Pimm *et al.* 1995; Lawton & May 1995; Leakey & Lewin 1995; Wilson 2002). Thomas *et al.* (2004) estimate that between 15 and 37% of species could be 'committed to extinction' by 2050.

Until recently, the dominant paradigm was that the processes and patterns of the biosphere were primarily determined by variability in the abiotic environment. Today it is recognised that life itself is critical in creating the planetary environment and keeping it within habitable limits (Lovelock 1979; Steffen *et al.* 2004). Loss of biological diversity, or the variability among living organisms, therefore has important implications for human well-being. The loss of biodiversity includes a reduction in the extent or condition of ecosystems, declines in the abundance or distribution of populations of individual species, or loss of genetic diversity within populations (CBD 2003a).

Since most ecosystem functions rely on the presence of particular organisms (accepting that several different species may have the capacity to deliver the same function), and the level of the function or service depends on the abundance of the organisms, a loss of biodiversity impacts negatively on the functioning of ecosystems and the provision of ecosystem services, or the benefits humans derive from nature (MA 2005a). For instance, our diet is dependent on biological variation: it is estimated that approximately 7000 plant species have been collected or cultivated for consumption (FAO 1998). Biodiversity plays an important role in the provision of economically-important ecosystem services such as protection against soil degradation and erosion, the reliable supply of high quality drinking water, the stabilization of the climate and the pollination of flowering plants. Biodiversity additionally contributes to ecosystem resilience, which buffers the impact of events such as floods and droughts, and will become increasingly important, particularly in poor communities, as climate and other environmental conditions become more variable and unpredictable (MA 2005a). People from all walks of life value biodiversity for spiritual and recreational reasons, as well as valuing its mere existence. For example, many countries have passed legislation to protect endangered species based on the belief that these species have a right to exist, even if their protection results in net economic costs. Independent of its use and non-use contribution to human well-being, biodiversity has intrinsic value in and for itself (MA 2003).

The importance of biodiversity in improving and sustaining human well-being led the nations of the world to set the target of reducing the rate of biodiversity loss by 2010 at global, regional and national levels. This target was originally set by the Convention on Biological Diversity (CBD), and endorsed at the World Summit on Sustainable Development held in Johannesburg, South Africa in 2002 (UN 2002). Given that the dominant drivers of biodiversity loss, namely changes in land-use, climate and nitrogen deposition, will continue to increase over this period (MA

2005a), stemming the loss of biodiversity will be a formidable challenge. At the same time, no scientific consensus exists with respect to measures for assessing the current state of biodiversity or monitoring progress towards this target (Royal Society 2003), although the CBD has agreed at a political level on a partial set of indicators for this purpose (CBD 2003c). The lack of scientific consensus on measures of biodiversity status, together with the increasing trends in the factors driving biodiversity loss, severely compromise the ability of society to take concrete action towards limiting the impacts and risks posed by this critical aspect of global change.

1.2 Environmental assessments

The increasing technical complexity and political sensitivity of emerging issues, particularly with respect to the environment, have prompted the inclusion of assessments as part of the decision-making cycle. Assessments, defined as a critical evaluation of information for purposes of guiding decisions on complex public issues, are a relatively recent addition to the science landscape. The new knowledge generated by research programmes is often not accessible to policy-makers. Assessments provide a mechanism through which the detail and necessary contention within the research community can be reduced in order to communicate a consensus view to policy-makers in clear and actionable terms. In contrast to research programmes, assessments are characterised by the synthesis of existing knowledge, targeting of their outputs specifically at decision-makers, and encouraging clearly identified evaluations and judgements by technical experts where contradictory or inadequate information exists. The assessment process is usually interdisciplinary and multi-authored, and includes a rigorous and transparent peer and stakeholder review element.

Several large environmental assessments have been undertaken in the past two decades in response to the concerns raised by global environmental change. These include the Ozone Assessment[1], the Intergovernmental Panel on Climate Change[2]; and the Millennium Ecosystem Assessment[3]. A number of other processes, such as UNEP's *Global Environmental Outlook*[4] and

[1] www.unep.org/ozone

[2] www.ipcc.ch

[3] www.maweb.org

[4] www.unep.org/geo

Global Biodiversity Outlook[5], partially meet the abovementioned characteristics of assessments. Most of these initiatives have attempted to adopt an integrative approach, given the importance of interactions between the biophysical and socio-economic spheres in relation to global change. The majority have incorporated the following features: an assessment of the current state and trend of an environmental issue, such as biodiversity or climate; understanding the major drivers affecting change; understanding the impacts of change, particularly on human well-being; exploring future scenarios of change; and investigating policy alternatives for avoiding or adapting to undesirable outcomes.

Indicators are an important tool in assessment initiatives. Indicators are variables that represent and provide information about particular attributes of a system. They may be direct measures of a phenomenon of interest, but are often proxy measures. Indicators are used for assessing the state of the environment, differences between places or changes over time, possible trajectories of change, and the effect of policy interventions. Assessment and research programmes make use of a wide range of indicators, but generally have a handful of 'headline' or 'aggregate' indicators, which are intended to raise awareness by providing a high-level synthetic overview for the public and politicians. Examples are the Living Planet Index (Loh 2002) and the Environmental Sustainability Index (World Economic Forum et al. 2002). These are typically underpinned by a much larger set of indicators that attempt to provide comprehensive and detailed information for specific users and for deciding on appropriate courses of action. Useful indicators represent attributes of fundamental interest to decision-making, and simplify or summarize several important properties of the system. Indicators are selected for use based on the extent to which they are measurable, use data that is already available or can be obtained through monitoring, are cost-effective, and are accepted and understood by users (Gallopín 1997).

This study was linked to the global Millennium Ecosystem Assessment (MA) initiative. The objective of the MA was to provide an integrated assessment of the consequences of ecosystem change for human well-being. The MA focused specifically on ecosystem services, defined as the benefits which people obtain from ecosystems (MA 2003). These include provisioning services (sometimes called 'ecosystem goods') such as food and water; regulating services such as flood and disease control; cultural services such as spiritual and recreational benefits; and supporting services that maintain the conditions for life on Earth, such as nutrient cycling. The MA regards

[5] www.biodiv.org/gbo

biodiversity as an underlying condition necessary for the delivery of ecosystem services. In some cases, such as when it is the resource for nature-based tourism, it is a service in its own right.

A novel feature of the MA was that it was conducted at multiple spatial scales. Approximately 30 sub-global assessments were linked to the global-level assessment, each with their own user group and problem definition. The primary reasons for adopting a multi-scale approach were to

- permit reporting and response options to match the scales at which decision-making occurs, in order to enhance the uptake of the assessment findings;
- permit individual ecological and social processes to be assessed at the scales at which they operate, and linked to processes at different scales; and
- allow for cross-scale validation of the assessment findings.

The Southern African Millennium Ecosystem Assessment (SAfMA) was one of the sub-global assessments linked to the MA, and was itself conducted at three spatial scales (Biggs *et al.* 2004). At the broadest level was an assessment of mainland Africa south of the equator (Scholes & Biggs 2004). Nested within this regional-scale study were basin-scale studies of the Zambezi (Desanker *et al.* 2005) and Gariep (Bohensky *et al.* 2004), and nested within these basins were several local or community assessments (Lynam *et al.* 2004; Shackleton *et al.* 2004). The SAfMA participants decided at the outset that three core clusters of ecosystem services, namely water, food and biodiversity, would be assessed at all three scales in order to investigate the effect of scale on assessment outcomes. With respect to biodiversity, this raised two needs:

- a method of assessing the state of biodiversity in southern Africa,
- that could be applied at and compared between multiple spatial scales.

1.3 Assessments of biodiversity in southern Africa

Southern Africa has an extraordinarily rich biota relative to its size (Huntley 1989; Burgess *et al.* 2004). Furthermore, southern Africa's biodiversity, with some exceptions, is currently in better condition than in many parts of the world (Scholes & Biggs 2005; Hoekstra *et al.* 2005). This situation presents both an opportunity, and an ongoing threat. Paradoxically, it is also why Africans are under increasing pressure to conserve their biodiversity (Scholes *et al. in press*), and monitor changes in biodiversity over time.

Southern Africa's vertebrate fauna is particularly rich. Approximately a fifth of the world's 4700 mammal species occur in sub-Saharan Africa, including most of the world's remaining megafauna (Brooks *et al.* 2001). A similar fraction of the world's approximately 10 000 bird species occur in the region (Burgess *et al.* 2004). Biological variety is especially high in several centres of biodiversity. For instance, the Cape Floral Kingdom in South Africa has about 3% of the world's known higher plant species, despite occupying only 0.05% of the global land surface (Huntley 1989; Le Roux 2002). Another important centre of biodiversity in southern Africa is the Drakensberg Montane Grasslands and Forests (Küper *et al.* 2004), which includes the Maputoland-Pondoland-Albany phytochorian defined by White (1983) (Figure 1.1).

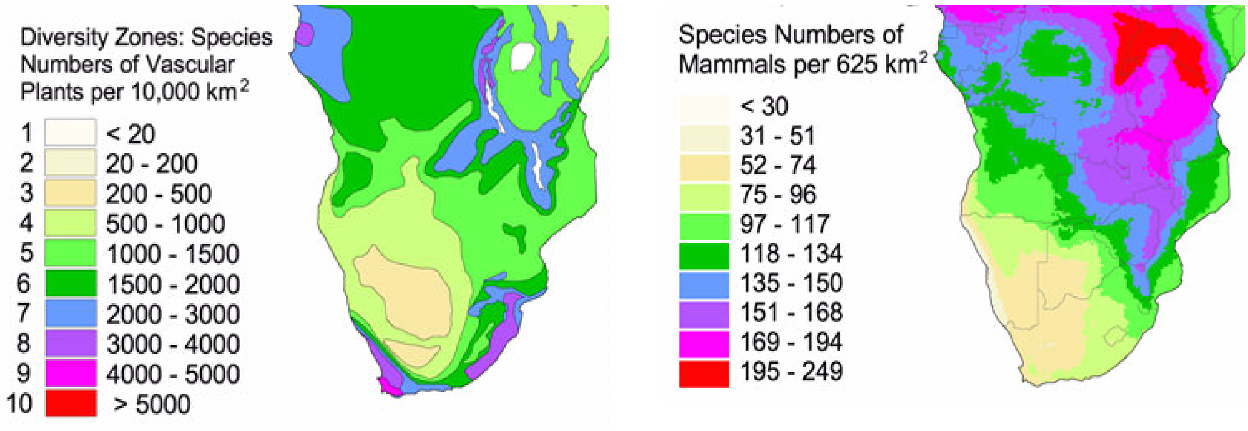

Figure 1.1 The distribution of biodiversity in southern Africa: a) Vascular plants (Mutke & Barthlott *in press*), and b) Mammals[6]. Adapted from Scholes *et al.* (*in press*).

The most widely used measures of biodiversity performance in southern Africa have been species extinction, and the fraction of land in protected areas. Neither of these measures adequately met the needs of SAfMA in terms of a method of assessing the state of biodiversity in southern Africa. Of the approximately 30 000 species of endemic plants (inferred from Huntley 1989) and 3 000 species of endemic vertebrates (Olson *et al.* 2001) in the region, 41 plants and 12 animals are believed to have become extinct in the last two centuries (IUCN 2002; Golding 2002). This does not adequately reflect the extent of the changes in land-cover and biodiversity that have occurred in southern Africa over this period (Scholes & Biggs 2004). Furthermore, extinction only indicates a change once it is too late to do anything about it.

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[6] Distribution maps for all non-marine mammal species were compiled as part of the IUCN Global Mammal Assessment (GMA) (www.iucn.org/themes/ssc/programs/gma/index.htm).

The extent of protected areas has been growing rapidly in southern Africa over the past few decades. Southern Africa currently has 7% of its land area in parks, and another 8% in formally demarcated sustainable use areas (Scholes & Biggs 2004). This exceeds the international guideline of 10% land area under formal protection, and in general, southern Africa is acclaimed for the quality of management of its parks. However, using protected area as the primary measure of biodiversity status ignores the 90% of the landscape where most of the biodiversity changes are occurring, and are likely to occur in future as the expansion of protected areas levels off.

Substantial effort has been invested in conservation priority analyses in southern Africa. There are two major approaches: representation and hotspot analyses. Both are aimed at identifying priorities for conservation action, rather than providing an integrated assessment of biodiversity status for comparison between different regions or scales. These analyses therefore did not meet the requirements of SAfMA, but are nevertheless briefly described here as they have had a significant influence on biodiversity planning and policy in southern Africa.

Representation analyses aim to identify a set of areas whose conservation would protect representative samples of all ecosystems. They may additionally include considerations of threat or vulnerability, and criteria such as cost of available land. Variations of these sophisticated systematic conservation planning techniques have been locally developed in South Africa and applied on a regional scale (e.g. Cowling and Pressey 2003). A recent study was carried out by Olson and Dinerstein (Olson & Dinerstein 2002), to identify priority areas at a global scale. Several of these areas lie in southern Africa, and many are densely inhabited, creating the potential for conservation conflict (Balmford *et al.* 2001). Hotspot analyses, as carried out by Conservation International, aim to highlight areas with high concentrations of endemic species that are undergoing exceptional loss of habitat (Myers *et al.* 2000). A complementary approach is to identify the remaining areas in each biome least impacted by human activity (Sanderson *et al.* 2002). At large scales, priority setting analyses, including Birdlife International's Endemic Bird Areas (Stattersfield *et al.* 1998) and the IUCN/WWF's Centres of Plant Diversity and Endemism (Davis *et al.* 1994), show significant overlap (Myers *et al.* 2000; WCMC 2000) (Figure 1.2).

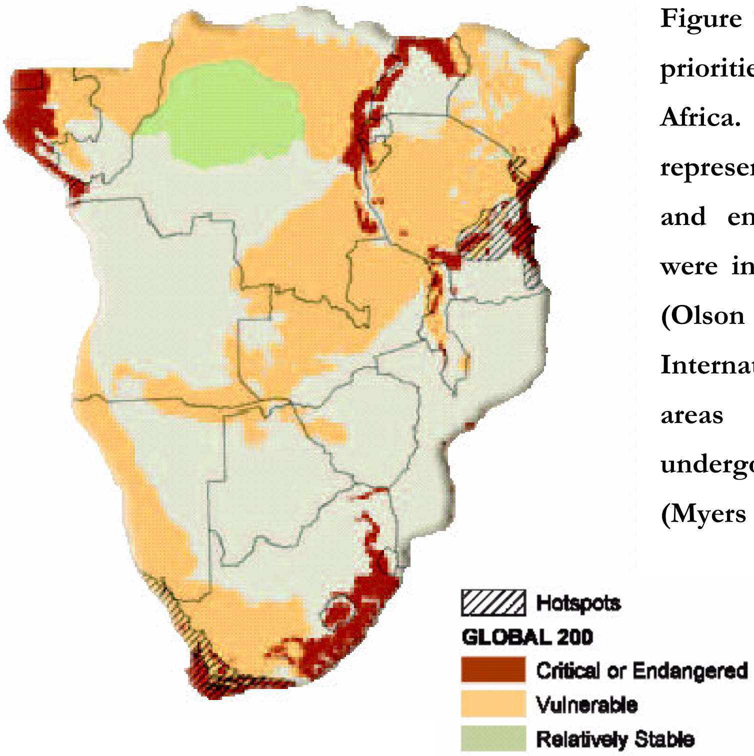

Figure 1.2 Areas that have been proposed as priorities for conservation action in southern Africa. The WWF Global 200 identifies representative examples of all ecosystems and emphasises biodiversity features that were in place before major human impacts (Olson & Dinerstein 2002). Conservation International's hotspot analyses highlight areas of high endemism which are undergoing exceptional loss of habitat (Myers *et al.* 2000).

1.4 Aim, scope and approach of this study

The aim of this study was to derive and apply a macro-ecological indicator to assess biodiversity status at multiple scales. In order to ensure relevance and usability, certain criteria were defined that the indicator was required to fulfil. These criteria expanded on those developed by the Convention on Biological Diversity (CBD 2003a; CBD 2003b; CBD 2003c):

- **Scientific soundness**. The indicator had to represent an actual observable quantity (e.g. number of species) that synthesized the impact of various factors impacting on biodiversity, and not simply an arbitrary value derived by a relative scoring and weighting of factors thought to relate to biodiversity status (e.g. as done in the Environmental Sustainability Index). The indicator had to incorporate and be sensitive to the main factors that are known to influence biodiversity status. It was required to take account of the graduated nature of the impact of different human activities, and not adopt a binary approach to classifying the landscape (e.g. protected, unprotected). Similarly, the indicator had to account for changes in the abundance of species or the extent of ecosystem types, and not simply their presence or absence. Extinction is the final step in a long process of reduction in the abundance of organisms or communities. The uncertainty in the index had to be quantifiable, and the main sources of uncertainty had to be identifiable.

- **Policy relevance**. The indicator had to be intuitively understandable and easy to communicate. In addition, the indicator was required to send an unambiguous message that could be the basis for policy and decision-making. For this reason, it needed to be highly synthetic (i.e. bringing together several linked information streams into a single, integrated indicator) and amenable to aggregation at the large spatial scales at which national and international policy is made. At the same time, it had to allow for disaggregation into its components and to sub-national scales in order to ensure transparency and meet the needs of users in different spheres of decision-making. The indicator had to be sensitive to changes at policy-relevant spatial and temporal scales. It had to enable comparison between a baseline situation and a policy target, and be useable in reconstructions of past trends and scenarios of the future. Furthermore, it needed to be in a form that could be related to concepts such as economic value and human well-being, so that trade-offs could be evaluated.

- **Applicability at multiple spatial scales**. It was essential that the indicator was amenable to application at, and direct comparison between, different spatial scales. For example, the value for a particular municipality should be directly comparable to that for a sub-continental region without complex rescaling procedures. In addition, the indicator needed to be quantifiable for areas either defined by ecological boundaries (e.g. biomes), or by political boundaries (e.g. provinces or counties).

- **Practicality and affordability**. It was important that the indicator be measurable with sufficient accuracy at an affordable cost, for both data-rich and data-poor regions. In addition, the indicator had to be able to make use of differing types and quality of data sources, as well as hybrid data sources.

This study introduces a new indicator, the Biodiversity Intactness Index (BII), as a measure which meets these criteria. It then demonstrates its application in several policy-relevant contexts. While the BII can conceptually be applied in any part of the world, including the oceans, and potentially inland water and coastal systems, the application of the index in this study was restricted to the land area of southern Africa, defined as the region covering South Africa, Lesotho, Swaziland, Namibia, Botswana, Zimbabwe and Mozambique. The study adopted an assessment approach, focusing on the synthesis of existing data, rather than on the collection of new data, and aiming to identify gaps in knowledge and highlight uncertainties. In the spirit of the MA, different types of verifiable knowledge sources were used. In this case, the primary data sources were existing species inventory data that had been collected though field surveys, satellite-derived land cover data, and the tacit knowledge of scientific experts.

1.5 Dissertation structure

This dissertation consists of eight chapters (Figure 1.3). Other than this chapter and the concluding chapter, they are all in the form of stand-alone scientific publications. There is thus an inevitable degree of repetition and redundancy between chapters. Chapters 2, 3 and 7 have already undergone scientific peer review and have been accepted for publication. Chapter 4 has been published online in the form of electronic conference proceedings. Chapter 5 has been submitted for review, while Chapter 6 is being finalised for submission. Chapters have not been organised in chronological order of publication, but according to a logical progression of ideas. All the stand-alone papers have multiple authors. Each is prefaced with a brief statement of the background to the paper, its publication status at the time of submission of this dissertation, and my specific contribution to the underpinning research and writing of the paper.

This chapter provides the broader context of global environmental change and assessments, and establishes the objectives of the study. Chapter 2 introduces the multi-dimensional nature of biodiversity, and provides a survey and assessment of existing biodiversity indicators. It serves as the literature review for this dissertation, and motivates the need for a macro-ecological indicator of biodiversity status that can be used for policy purposes.

Chapter 3 introduces, defines and describes the BII as a synthetic, policy-relevant indicator, specifically with regard to assessing progress towards the international target of reducing the rate of biodiversity loss by 2010. The algorithm and data used to calculate the index are described in detail in this chapter. To illustrate its application, the index is calculated for southern Africa for the year 2000 and the rate of biodiversity loss is estimated for the decade of the 1990s. In order to establish the scientific credibility of the index, Chapter 3 also presents summarized results of tests of the scalability and sensitivity of the index, and a validation of the use of information derived from expert judgment. Full details of the tests that were performed to verify the application of the BII at multiple policy-relevant scales are presented in Chapter 4. This chapter also details the analyses that were performed to test the sensitivity of the BII to land use and species richness data of varying quality and resolution.

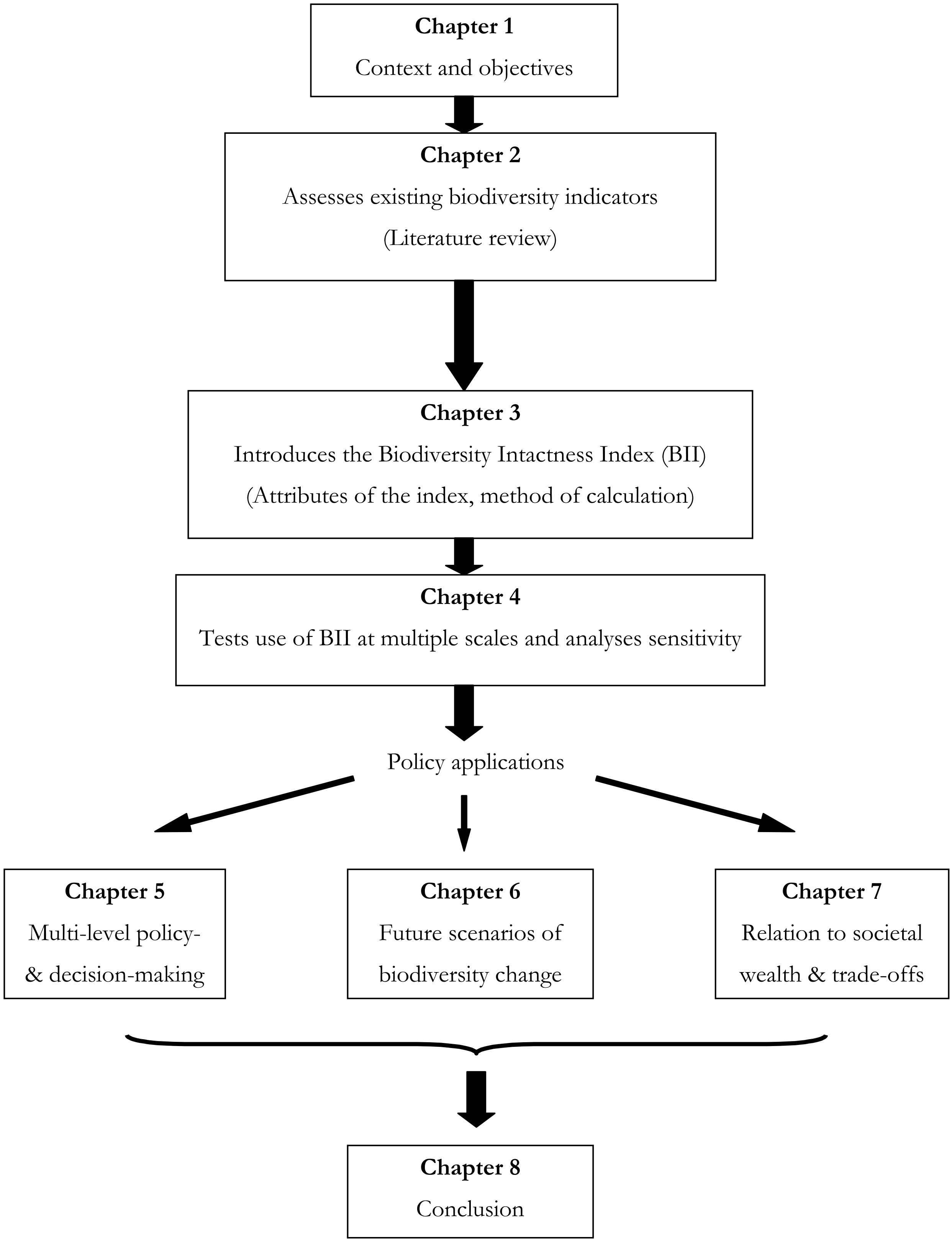

Figure 1.3 Organising framework for the dissertation, showing the primary purpose of each chapter and its role in the overall study. Note that Chapters 2 through 7 have been prepared as stand -alone publications; details are given on the cover page preceding each chapter.

Selected policy-relevant applications of the BII are explored in Chapters 5 through 7. The need for an indicator that can be used at multiple levels of decision-making was a key criterion in the derivation of the BII. Chapter 5 interprets the results of applying the index at a several policy-relevant spatial scales in South Africa, and explores the implications for biodiversity conservation in the country. Another important criterion was that the indicator could be used to explore the implications of different policy or management approaches. Chapter 6 makes use of the IMAGE land-cover model (Alcamo *et al.* 1998) and the four MA global scenarios (MA 2005b) to explore possible changes in biodiversity intactness in southern Africa over the 21st century. Trade-offs lie at the heart of policy- and decision-making. In Chapter 7, the BII is used as a proxy for renewable natural capital, in order to relate historic changes in biodiversity to total societal wealth. This enabled an exploration of the nature and extent of the trade-off that has been made between the preservation of biodiversity and the economic development of South Africa.

The final chapter (Chapter 8) draws conclusions with respect to the use of the BII as an indicator of biodiversity status. It highlights the major policy-relevant findings that have emerged from the application of BII in southern Africa, and identifies avenues for further research.

1.6 References

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Chapter 2: Biodiversity indicators

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This chapter introduces the multi-dimensional, multi-level nature of biodiversity and provides an assessment of biodiversity indicators currently in use. It serves as the literature review for this dissertation. The chapter was prepared by invitation as part of the *Assessment of Sustainability Indicators* (ASI) project[1], coordinated by the Scientific Committee on Problems of the Environment (SCOPE)[2], the International Human Dimensions Programme on Global Environmental Change (IHDP)[3], and the United Nations Environment Program (UNEP)[4].

Agenda 21, adopted at the 1992 United Nations Conference on Environment and Development in Rio de Janeiro, expressed the need to formulate sets of indicators in order to better monitor and foster sustainable development. Hundreds of different indicators have since been developed and are being used in diverse contexts. The aim of the ASI project was to provide a scientific assessment of existing indicators of sustainable development, focusing on the scientific validity of methodologies, relevance to sustainable development, policy relevance, and the feasibility of different indicators. This chapter specifically addresses indicators of biodiversity, and forms part of the SCOPE volume that synthesises the outputs of the ASI project. In the SCOPE volume, the chapter is preceded by an introductory chapter that explores the concepts of sustainability and indicators in detail, hence they are not elaborated on in this chapter. Prior to acceptance for publication, the chapter was reviewed by two experts, overseen by a review editor.

The chapter was authored by myself, R.J. Scholes, B. ten Brink and D. Vačkář. The base text was developed by myself and R.J. Scholes, and circulated to the other authors for additions and comments. I was responsible for coordinating the writing and submission of the chapter, as well as for incorporating the comments of the co-authors, reviewers and editors. My leading role is recognised by the authors in the order in which the authors are listed.

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Chapter 2: Biodiversity indicators

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2.1 Introduction

Human actions compromise the information content of the biosphere, which is contained in its biological diversity (biodiversity, for short). The fundamental logic of biodiversity conservation is that the variety of living things matters for a range of utilitarian (human-centred) as well as intrinsic reasons. Variation is the raw material of evolution and the source of novel and useful biological products. An ecosystem composed of very similar organisms will react differently to imposed stresses than one composed of dissimilar organisms, although general predictive rules remain elusive. Biodiversity has aesthetic appeal in all cultures and many people share a moral imperative to conserve a representative sample of the full range of biological variation. In short, it is widely agreed that all else being equal, more diversity is better than less, especially in the context of natural or self-regenerating ecosystems, but the critical limits are unknown.

Many biodiversity indicators have been proposed (Reid *et al.* 1993; Delbaere 2002b; CBD 2003c; CBD 2003e), but as yet none have been universally accepted and applied (Royal Society 2003). The adoption by the World Summit on Sustainable Development and the Convention on Biological Diversity (CBD) of a target to reduce the rate of biodiversity loss by the year 2010 (CBD 2003a; CBD 2003d) has accentuated the need to achieve convergence on this issue. As a result, the international discussion on indicators has been accelerated and preliminary agreement on five trial indicators reached within the CBD (2003b).

The difficulties in establishing operational indicators of biodiversity stem from three main sources: firstly, inadequacy of much of the data; secondly, the loss of information that occurs when a complex and multi-dimensioned concept is reduced to a one dimensional indicator; and thirdly, our rudimentary understanding of the causal links between biodiversity and ecosystem function. This chapter presents the basic concepts important to monitoring biodiversity, provides a broad overview of current developments in biodiversity indicators, and outlines future possibilities in this field. We focus on indicators at the national to global levels, in order to support policy decisions related to sustainable development.

2.2 The theoretical basis for biodiversity definitions

A scientific definition of biodiversity might be 'the *complexity* of living systems at all organisation levels'. Many definitions are valid within the context of their specific use, and no simple definition can cover all aspects: natural *versus* human-altered diversity; evenness *versus* richness; the various spatial (α, β and γ biodiversity) and temporal dimensions (phylogenetic biodiversity); and the biological incompatibilities of increasing diversity at all organisation levels simultaneously. Because of the broad definition, it is very difficult to derive verifiable targets, measures and measurements of biodiversity at this time. Which biodiversity should be conserved at the expense of which other biodiversity?

The conceptual framework attributed to Noss (1990) is a useful way to organize the various inter-related facets of biodiversity (Figure 2.1). It proposes that biodiversity is expressed at three main *levels of organisation* (ecosystem, organism and gene) and in three *aspects* (compositional, structural and functional). Thus, focusing for instance at the organism level, a sample composed of several different species is more diverse than one with fewer species. Even with only one species, a sample that includes both big and small individuals, perhaps organized into clumps, is structurally more diverse than one where all the individuals are the same size, organized in a perfect grid. If several species were present but all did exactly the same thing, functionally they would be less diverse than a sample which included organisms that had very different roles: plants, herbivores, carnivores and decomposers, for instance.

The same three aspects of biodiversity can be defined at the supra-organism scale (the ecosystem) and the sub-organism scale (the gene). A landscape that is a mixture of forest, grassland and cropland is compositionally, structurally and functionally more diverse than one which is forest only. A cloned crop in which each individual is identical to every other is less diverse than a traditional land-race that contains genetic variation. On land, ecosystems are mapped almost entirely based on the distribution of relatively easily identified plant structural formations, such as forests or grasslands. Structure is often closely linked to function; the biggest potential divergence is between composition and function. For instance, the microclimate in a forest is related more closely to the relative proportion of tall, medium and short plants, than to the variety of species present.

The fundamental evolutionary process generating biodiversity is mutation, and its stabilisation within populations by speciation. This is a slow and still poorly understood process. Ultimate biodiversity loss occurs by extinction (of specific genes, species or ecosystems), which occurs at an unsteady and only roughly quantified rate, even in the absence of human threats. It is nevertheless apparent that the Earth is currently in a period of net biodiversity loss (Leakey & Lewin 1995).

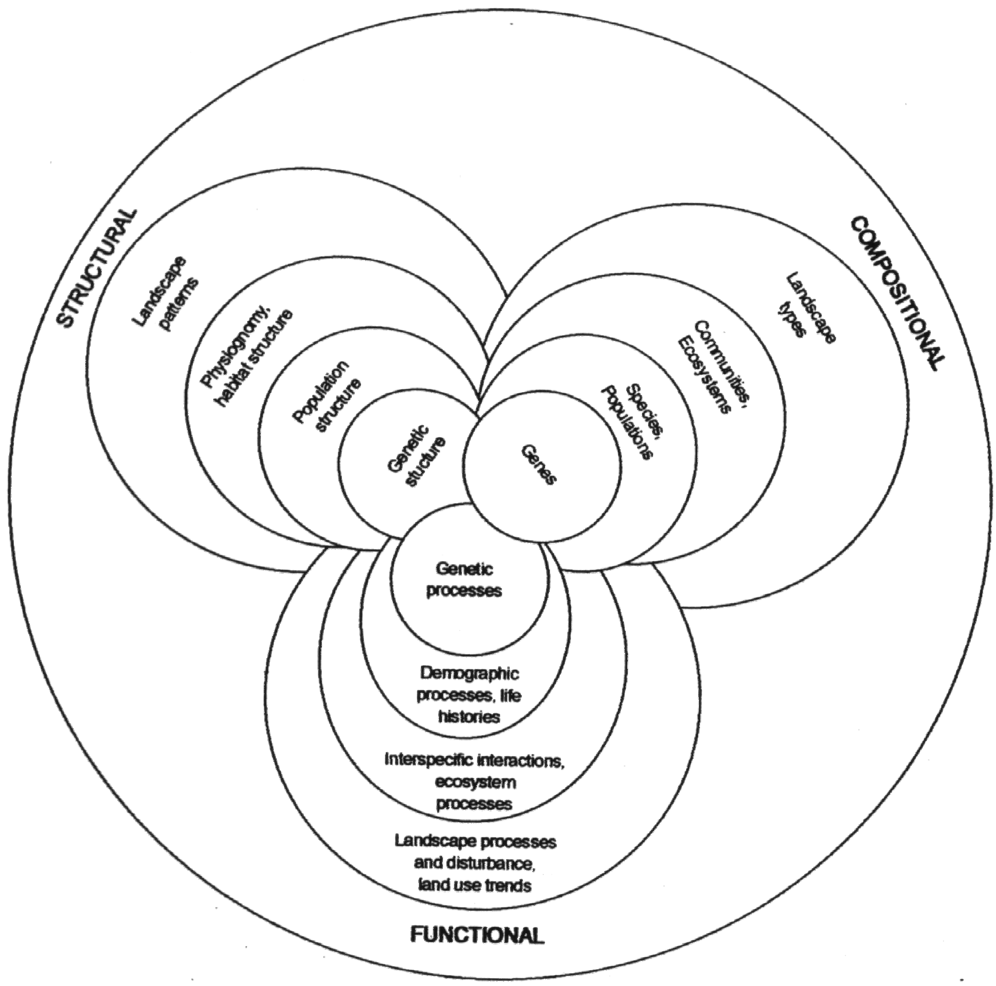

Figure 2.1 Compositional, structural and functional attributes of biodiversity at four levels of organisation (Noss 1990).

Biodiversity loss, however, entails much more than extinction, and occurs at all levels of organization (CBD 2003a; CBD 2003b; CBD 2003d). At the ecosystem level it consists of a reduction in the extent, condition or productivity of ecosystems; at the organism level a decline in the abundance (ultimately extinction), distribution or sustainable use of species populations; and at the gene level it comprises loss of gene-level diversity within populations (genetic erosion). Anthropogenically-driven biodiversity loss manifests as a decrease in abundance of many species, and an increase in a few, leading to homogeneity at multiple scales.

Absolute biodiversity loss has to be measured relative to some ***baseline*** state, whose choice raises both political and practical concerns. A baseline state is not required in order to determine relative biodiversity loss, but at least two measurements in time are needed. Four baselines have been proposed for use by the CBD (2003e):

- Before any human interference (which is both illogical and unfeasible for many long-inhabited parts of the world);
- Before major interference by industrial society (recommended as most appropriate);
- From the time the CBD entered into force i.e. 1993 (results in a bias towards developed countries);
- Extinction threat according to the IUCN Red List categories.

Other possible baselines include viable population levels, species richness, a specific reference year, maximum sustainable yield levels, or a defined 'desired state'.

Richness and evenness are another set of ideas widely used in biodiversity observation. They have their origin in information theory (Pielou 1969). Information diversity indices combine the number of classes with the proportions in each particular class. Richness is a count of how many different varieties (classes) exist in a given sample. Evenness measures the degree of dominance among the different varieties. Many ecological diversity indices have been proposed (Magurran 2004). Interpretation of these indices is rather complicated, and a range of approaches has been suggested (French 1994; Iszák and Papp 2000). Such indicators do not distinguish between native, introduced and cultivated species, which is often desired by policy makers. A 'native' species, as opposed to an 'introduced' species, is one that occurs naturally in a particular area. Cultivated species have generally been specifically bred for purposes of cultivation. A minimum description of diversity requires *both* richness and evenness concepts, in the same way that an analysis of poverty usually includes measures both of absolute income and measures of income inequality.

α, β and γ biodiversity. α biodiversity is the number of types within a homogeneous patch, usually a very small area. γ biodiversity measures diversity at a more regional scale. β biodiversity measures the species turnover between adjacent ecosystems. α biodiversity can grow due to introduced species while γ and β biodiversity decline because the systems share an increasing set of common species, the so-called homogenisation process.

States, drivers and proxies. As in all indicator systems, it is useful to distinguish between indicators of the state of biodiversity, and indicators of those things causing changes in the state ('drivers', such as harvest pressure, pollution, fragmentation) and responses to the state (such as the area under formal protection). Because of data deficiencies, we are often forced to use proxies and surrogates as indicators of state and drivers. Exhaustive surveys in highly diverse regions require a great deal of time and money. Therefore simpler surrogates for mapping diversity, such as higher-taxon richness, or the richness of indicator taxon groups have been proposed and tested (Williams *et al.* 1998), with partial success. Bioindicator approaches measure the abundance of sensitive species ('sentinels' or 'detectors') or genetic markers in assessing ecosystem health, but are not measures of biodiversity *per se*.

2.3 Biodiversity indicators that have been proposed

Several hundred indicators to measure the status of biodiversity have been proposed or are under development. There exist indicators for all the levels of biodiversity (genetic, species, and ecosystem), and all the aspects of biodiversity (composition, structure, and function), but the amount of effort invested in these different areas has been uneven.

2.3.1 Gene-level indicators

At the sub-organism level, diversity can be measured by 'richness' type counts of the number of different varieties within a species (for example, the number of land-races of one crop) or the number of alleles present in a meta-population. These measures should be accompanied by 'evenness' type measures of the relative dominance of the varieties.

Genetic biodiversity can be estimated by measuring appropriate molecular markers. Most studies on genetic variation within species are based on random markers. Current work is investigating the use of markers targeted at genes exhibiting ecologically relevant activity (Tienderen *et al.* 2002). Indices of relatedness, whether they are based on cladistics (Clark and Warwick 1998) or genetic distance (Nei 1972; Takezaki and Nei 1996), have potential for measuring gene-level diversity both between and within species. They have the advantage of not requiring the actual genes to be exhaustively inventoried. Examples are the measures that have been used to try to estimate soil microbial biodiversity (Pankhurst *et al.* 1997) when the species are unknown (and the species concept may not even apply).

Despite very rapid developments in this field, it is our assessment that robust and widely applied gene-level diversity indicators are still a decade off. Until then, the species, imperfect as it is, is the most robust scientific basis for constructing most indicators.

2.3.2 Species-based indicators

A conceptual problem that faces all species-based indicators is the question: do all species matter equally? If not, on what basis is the weighting assigned? Where, for practical reasons, a subset of species is used, can they be assumed to represent the unsampled species?

Species Richness (i.e., the number of recognized and recorded species within a given set of taxonomic groups and within a given bounded area) is the simplest and most widely-used biodiversity index. The data are derived from collection labels in museums or field observations, and may be extrapolated over unsampled locations to create inferred species distribution maps, using either explicit techniques such as habitat modelling using GLIM or GARP (Crawley 1993; Stockwell 1994), or expert judgement. The principal problems with species richness as a biodiversity indicator are:

- It is critically dependent on the quality and completeness of collection and classification. Changes in richness are often simply a result of taxonomic revisions or more thorough observations, rather than changes in the underlying biodiversity. Some stability has been achieved in richness numbers for 'well known' taxa, such as birds, mammals and some plant groups (Govaerts 2001; Bramwell 2002; Scotland and Wortley 2003) in well-studied areas, but for the vast majority of the biological realm, less than half of the postulated extant species have been formally described (Groombridge & Jenkins 2002). In some parts of the world, even plant and vertebrate observations are extremely incomplete (Prance *et al.* 2002).

- Richness is strongly spatially dependent. Typically, the species richness within an ecological region rises to an asymptote in relation to the area sampled (the 'species-area curve'). The shape of the curve varies between ecological regions and taxa, and can itself be used as a biodiversity index (Cowling *et al.* 1989). Comparing species richness between locations without taking this relation into account is misleading. Special procedures, such as rarefaction (Hurlbert 1971) correct for biases related to unequal sampling area.
- Species richness is a very insensitive indicator of biodiversity loss. It provides no indication of changes in the abundance of component species in a community, or of changes in their phylogenetic and functional diversity traits. A decrease in richness only occurs through extinction. *Extinction* is the loss throughout the world of a species or variety, while *extirpation* is the loss of a species or variety within a portion of its range.
- Species richness *per se* does not distinguish between native and introduced species. In disturbed areas total species richness may increase due to introduced species, while populations of native species are reduced but not entirely extirpated. In this case species richness may provide perverse signals. It is therefore recommended (CGER 2000) that counts of introduced species be kept separate from those of native species. Maximising richness, by introducing alien species, is seen as a perverse objective.

Extinction, or the risk of it, has been widely used as a measure of biodiversity loss (e.g. IUCN 2002). It has the advantages of being extremely easily grasped and compelling, but as a practical biodiversity management indicator has several drawbacks:

- It is surprisingly difficult to prove that a species is extinct. How do you know that it is not just hiding?
- If a species is not known to science, its extinction is invisible.
- By the time a species has gone extinct (and been shown to be so) it is far too late to do anything about it. For this reason, the IUCN creates a variety of levels of 'threat' which are indicators of the last phase of the lengthy process of biodiversity loss.
- Species extinction is a natural process, balancing, in the long term, the process of speciation. What rate of extinction is clearly too high?

Endemism is a widely-applied refinement on species richness. This is the number of species found only within a specified area, and nowhere else on earth (in the wild). Endemic richness, at the species or higher taxonomic level, and sometimes in conjunction with an assessment of threat, is often used as an indicator of 'biodiversity hotspots' (Reid *et al.* 1993; Williams *et al.* 1998; Myers *et al.* 2000). Obviously, the loss of an endemic species within its entire range is more critical than the local loss of an otherwise widespread species. Endemism, however, suffers the same problems as species richness.

Complementarity measures. A sophisticated set of indices has been developed by conservation biologists for the purposes of optimising the design of networks of protected areas (Margules and Pressey 2000). These measures are based on quantitative targets that specify how much of the landscape is needed to conserve a representative set of sample species (or sometimes habitats). They may, additionally, include considerations of threat or vulnerability.

2.3.3 Population abundance-based indicators

These indicators are based on trends in the abundance (i.e. number, biomass or cover) of individuals within a target population, which may be the total global population for that species, or some well-defined part of it. Abundance-based indicators provide the advance warning of impending loss that richness-based indicators do not, since the underlying information is continuous rather than binary (present/absent). Such indicators are therefore more sensitive and useful for policy purposes than richness-based indicators, particularly for setting policy targets.

Hughes *et al.* (1997) estimated that there are on average about 200 separate populations per species. Because of the amount of effort required to get reliable, repeated abundance estimates of wild species from just one population, such censuses are restricted to a tiny fraction of all known species. They are often either 'charismatic' species such as elephants, whales, and migrating waterfowl, or 'useful' species such as fish, timber or domestic animal stocks. The WWF marine ecosystem indicators are an attempt to apply population measures with a rapid, standardised sampling protocol (Wilkinson 2000). The Living Planet Index (Loh 2002) and species trend indices (e.g. Gregory *et al.* 2002) are a population approach based on a relatively small set of species selected to represent the major groups. The Natural Capital Index (CBD 1997; Ten Brink *et al.* 2002; CBD 2003c) and the Biodiversity Intactness Index (Scholes and Biggs 2005) (see section 2.6) are in principle both population-based measures.

Changes in species populations may be correlated with changes in diversity at the genetic level. The smaller the population, the smaller its genetic variation, particularly if losses are concentrated around the edges of the species' area of distribution.

Not all species have to be measured to indicate the overall process of biodiversity loss. A representative sample would be sufficient, and a deliberately-selected set of indicators may be even more sensitive if the selection was well-founded. Population-based indicators are the focus for much of the current activity in biodiversity indicator development.

2.3.4 Phylogenetic and evolutionary indicators

The fundamental unit of biodiversity is an evolutionary one (Faith 2002). Santini and Angulo (2001) propose an index for the estimation of evolutionary potential, which they define as the potential of a member of the genealogical hierarchy to persist for ecologically significant periods of time. It is the balance between diversification and extinction of the evolutionary unit considered.

2.3.5 Area-based indicators

These can conceptually be considered as abundance-based measures at the ecosystem level, corresponding to population measures at the organism level. They typically express the area (km^2) at a particular time of a defined ecosystem. The area may be expressed as a fraction of some reference state, such as the supposed original ('potential') area. A well-known example is the Global Forest Resources Assessment (FAO 2001). Others are assessments of coral reefs (Wilkinson 2000) and mangroves (FAO 2003). Biodiversity, at all levels, needs an area within which to exist. When half the habitat of a given species is lost, the abundance of that species is more-or-less halved. This makes area an easily understood and easily measured indicator.

Fragmentation indicators are theoretically based on concepts of island biogeography, which predict that as fragments are isolated from a larger mass, they will lose species. They vary from the simple (e.g. mean fragment size, perimeter length to bounded area ratio) to the sophisticated (fractal-based indices). Road network density has been used as a proxy for fragmentation.

Area-based indicators are built on readily observable structural features, such as the cover by trees or coral reefs and thus are typically at the highest level of ecosystem classification, the biome. In principle they are straightforward and uncontroversial, but in practice it is hard to ensure uniformity of application of the definition. As a result, the variation between different sources of information (compare FAO (2001) to Achard *et al.* (2002), for instance) is usually much higher than the temporal trend, at least in the short term. However, the relative temporal trend is usually consistent in direction, if not magnitude. Only a few time series of trends in biome extent exist at the global level (Jenkins *et al.* 2003).

It is not clear what the equivalent boundaries for oceanic ecosystems are, since their edges would be expected to be naturally somewhat more variable in time. The Large Marine Ecosystems (Sherman *et al.* 1990) have fixed spatial definitions, and are thus not useful as indicators by themselves (though the fraction degraded within them would be useful). The Marine Biogeochemical Provinces (Longhurst 1998) are largely mapped using remote-sensing of sea-surface temperature and chlorophyll content, and are highly dynamic over period of weeks, making them too variable to be useful in the short term. Marine ecosystems may therefore lend themselves better to indicators at the species or community (e.g. seagrass, coral reefs) level.

2.3.6 Functional indicators

Indicators of functional biodiversity are least developed. There are many theories predicting a positive relationship between compositional biodiversity and functional attributes, but the generality and form of the link remains contested. It is unclear whether the link rests on biodiversity in general, or the presence of specific functional groups, or niche complementarity (Loreau *et al.* 2002). The debate on the functional link between biodiversity and ecosystem net primary productivity is converging on a conclusion that a weak positive link exists, which tends to level off at relatively modest levels of plant diversity (in the order of 10 species) (Kinzig *et al.* 2001; Naeem 2002).

For policy purposes it may be pragmatic to view function as synonymous with the capacity of ecosystems to supply ecosystem services, briefly defined as 'the benefits that people obtain from ecosystems' (MA 2003). For example, the capacity to supply water, fish, timber, food, or carbon storage services can, amongst others, be regarded as indicators of ecosystem function. Data on services often exist, but their link to bio*diversity* remains obscure. In many cases, an equivalent quantum of service could be supplied by a less diverse ecosystem, though perhaps with less resilience.

The Millennium Ecosystem Assessment (MA 2003) makes a useful distinction between biodiversity as an ecosystem service in its own right, and biodiversity as an underlying condition necessary for the delivery of many other services. It is easy to confuse natural resources with biodiversity. For example, almost all the food that we eat has its origin in a plant or animal, which are in turn a component of biodiversity. But it is generally not the *diversity* of that source that is uppermost in our mind when we think of the nutritional service. Similar nutrition could be, and increasingly is, provided from a very small variety of sources. There is nevertheless a key role for biodiversity in food supply: it provides the source of variation needed to adapt the crop to a changing environment, and to reduce the risk of catastrophic failure in the event of a widespread stress (such as drought or disease). The case for a close connection between biodiversity and human well-being is more direct for the 'regulating' ecosystem services, such as control of pests and diseases.

While monetary measures of 'Natural Capital' (De Groot 1992; Constanza *et al.* 1997), which directly measure biodiversity function in monetary terms, are similarly of limited utility as biodiversity proxies, their ability to link to social and economic drivers and human use is a valuable attribute.

Non human-centred measures of ecosystem function usually boil down to direct or indirect measures of biogeochemical cycling, such as net primary productivity or evapotranspiration. The greenness of the land or ocean surface, and its spatial and temporal variability, are proxies for this type of measure that lend themselves to remote sensing.

At the organism level, species can be reclassified into 'functional types' or 'guilds': groups of species that share, to some degree, attributes such as basic physiology, reproductive strategy, trophic position and response to stress or disturbance (Smith & Schugart 1996). Richness and evenness measures can then be applied to functional types rather than species. The problem is that there is no standard for the definition of functional types, which can be endlessly subdivided until species or even lower basic units are reached.

2.3.7 Integrity indicators

Ecological integrity is the capacity to maintain a balanced, adaptive community of species having species composition, diversity and functional organisation comparable to that of natural habitats in the particular region. An example of an indicator is the Index of Biotic Integrity (Karr 2002), which scores a range of aquatic community parameters against defined standards. Terrestrial versions have also been proposed (Andreason *et al.* 2001). Some integrity indices are based on the levels of connectedness and self-organisation in ecosystems (Kutsch *et al.* 2001). 'Ecological health' concepts focus on assessment of the functional aspects of an ecosystem, without necessarily referencing them to a 'natural state'.

2.3.8 Inferential and composite indicators

There is a great temptation to reduce the complexity by combining a range of indices into a single measurement. The 'Human Influence Index' (Sanderson *et al.* 2002) is an example. Strictly, doing this by summation or averaging only makes mathematical sense if the individual indices have the same units and measurement scales. Very many social and economic indicators (such as the Human Development Index, (UNDP 2003)) are of such a multi-factoral nature. Sometimes the problem of incommensurability is addressed either by normalising all the numbers to some benchmark, or weighting the various components (in which case the weightings are implicitly unit conversion factors). Axis scores in Principle Component Analysis (or similar techniques) fall into this category. Multiplicative indices may make more mathematical sense, but should be based on a sound conceptual model, or else they will make no ecological sense.

Given the complexity and variety of biodiversity in and between ecosystems, composite indicators are probably one of the few ways in which information can be made digestible and understandable for politicians and the public (CBD 2003a; CBD 2003c). However, such indicators must be transparent about the way the various factors are transformed and combined. It must be possible to delve down into the individual components that make up the index, and to disaggregate them spatially.

2.4 Biodiversity indicators in use

Only a fraction of the hundreds of biodiversity indicators that have been proposed have been implemented on a regular basis (Delbaere 2002a; CBD 2003c). The most commonly used indicators are species-richness, number of threatened and extinct species, number of endemic species, trends in abundance of particular species, area loss of ecosystems, and the percentage of area protected and its derivatives. Some of these indicators are insensitive, provide perverse messages, or are not an indicator of state but of response. A list of biodiversity indicators currently in use and proposed under the CBD are given in Tables 2.1 and 2.2.

Many biodiversity measures had their origins in the taxonomic disciplines, which continue to have a strong influence through the emphasis on creating complete and correct lists of species: hence efforts such as the *Global Biodiversity Information Facility*[1] and *Species 2000*[2]. The rationalization that such initiatives bring is welcome, but taxonomic completeness, if achievable at all, is still many decades down the road. Given the indications that the current rate of extinction is abnormally high, urgency is paramount, and robust and scientifically defensible shortcuts are needed. Absolute taxonomic completeness is not necessary to measure biodiversity loss. Sampling a limited number of well-described species can provide sufficient information to guide interventions.

The main conservation advocacy groups have relied on the perceived threat of extinction, captured in Red Data Lists, since these do not require complete species inventories. 'Flagship species', in other words those with high public appeal, have attracted a disproportionate amount of attention. The policy focus is shifting towards an 'ecosystem-based' approach, which is intended to be more holistic. The prioritisation for protection of 'hotspots' and large untransformed areas at the ecosystem scale, based on indirect measures or expert-judgement, is a trend of the last decade.

The most recent trend is towards abundance-based indices. They have circumvented the data limitations by one or more of the following strategies: a) focussing on a small number of well-studied species; b) using models to supplement inadequate data series or c) using expert judgement in place of population censuses.

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[1] http://www.gbif.org/GBIF_org/what_is_gbif

[2] http://www.sp2000.org

Table 2.1 Single indicators (one variable related to reference value) currently in use (CDB 2003a).

TYPE	INDICATOR (not exhaustive)	QUESTION
I. State & Trend		
Ecosystem	◆ Area of ecosystem type e.g. forest, agriculture, built-up land	*How much natural area remains, how much is agricultural, and how much is built-up?*
	◆ Hotspots (areas of high endemism experiencing high human impact)	*Which ecosystems with high diversity of endemic species are threatened?*
Species	◆ Trends in representative species, particular taxonomic groups, exploited species, endemic species, migratory species, red list species, etc.	*What is the quality of the remaining natural area and agricultural area? What are the trends at species level?*
	◆ Percentage of threatened and extinct species in particular groups	*Which species are threatened?*
Structure	◆ Trends in structure variables e.g. canopy cover, ratio of dead to living wood, % area vital coral reefs	*What are the trends of ecosystem structures?*
Genes	◆ Number and share of livestock breeds and agricultural plant varieties ◆ Number of endangered varieties of livestock breeds and agricultural crops ◆ Share of major varieties in total production for individual crops	*Which genetic resources are threatened?*
◆ **II. Pressures/ Threats**		
Physical	◆ Annual conversion of self-generating area as % of remaining area ◆ Change in mean temperature and precipitation ◆ Road density ◆ Damming & canalisation of rivers ◆ Fire	*What is the size of the pressure, and its trend?*
Chemical	◆ Acid deposition ◆ P or N deposition ◆ Exceeding soil, water or air standards for particular pollutants	
Biological	◆ Total number of invasive species ◆ Total amount harvested per species	
Indirect	◆ Human population density ◆ GNP	*What factors influence the direct pressures?*
III. Use		
Provisioning	◆ Total amount harvested per species or species group	*What is the use?* *Is it sustainable?*
	◆ Per capita wood consumption	*How many people depend on the system?*
Regulating	◆ Carbon stored within forests	*What is the contribution to GDP?*
Cultural	◆ Total revenue from ecotourism	

IV. Response	
Legislation	◆ Total number of protected species as % of particular groups ◆ % Protected area by IUCN category
Targets	◆ NBSAP objectives met
Expenditure	◆ Expenditure of abatement & nature management measures (US$)
Management	◆ Number of protected areas with management plan ◆ Number of threatened & invasive species with a management plan ◆ Effectiveness of protection measures in protected areas
V. Capacity	
Personnel	◆ Nature research capacity in number of people ◆ Conservation policy capacity in number of people ◆ Nature site management in number of people
Legislation	◆ Number of physical and chemical standards
Monitoring	◆ Number of physical, chemical and biological variables measured ◆ Local site support groups and number of volunteer monitors

Table 2.2 Composite indicators that are currently in use (based on CBD 2003a).

Group	Indicator	Source
General state	• Natural Capital Index • Wilderness • Living Planet Index • Last of the Wild • Biodiversity Intactness Index	Ten Brink *et al.* (2002) Conservation International Loh (2002) Sanderson *et al.* (2002) Scholes and Biggs (2005)
Trends of components	• Species Assemblage Trend Indices e.g. Bird Headline Indicator, Living Planet Index	e.g. Gregory *et al.* (2002), Loh (2002)
Threat	• Red List Indicators on species groups • Hot spots • Human Footprint	IUCN (2002) Myers *et al.* (2000) Sanderson *et al.* (2002)
Pressures	• Total Pressure Index • Habitat-species matrix (Agricultural practices)	UNEP (2002)
Uses	• Sustainability of total use	
Responses	• Effectiveness of environmental measures • Effectiveness of area protection • Effectiveness of site management	

In October 2003, the CBD accepted a working paper that proposes a series of 18 indicators for application at national scale (CBD 2003d). They include measures at all three levels (ecosystem, organism, genetic) and all aspects (composition, structure and function), but are not intended to be comprehensive or integrated. They are the result of a consultation process that began around the time of the Global Biodiversity Assessment (1995).

Because of the lack of data on biodiversity trends, there is a tendency to use indicators of pressure ('drivers') instead. The Geobiosphere Load (Moldan 1997) and Ecological Footprint (Wackernagel *et al.* 2002) are indicators of pressure. Some pressures, such as population density or household dynamics, are relatively easily quantified (Liu *et al.* 2003). It is generally recognised that human-dominated landscapes are species poor, or invaded by alien species (Araújo 2003). Human appropriation of photosynthesis products (Vitousek *et al.* 1986; Rojstaczer *et al.* 2001) reduces the energy available to support wild populations and ecosystems.

2.5 Data issues

The key issue in the applicability of indicators is access to reliable and consistent information, particularly when attempting to apply indices at continental or global scales. The quality of knowledge varies greatly across biological groups: for instance from very good for birds, to very poor for soil microbes. This does not reflect their relative ecological importance, but their 'charisma' and ease of study. This unevenness of observation has geographical consequences: the tropics and the oceans are less well inventoried than the temperate land masses, because they have a greater biodiversity and a shorter record of scientific study. To an extent, broadening the information sources to include traditional or indigenous knowledge can help alleviate the problem for the more prominent groups, but it is unreasonable to expect indigenous people to have knowledge about subjects that may not have seemed necessary or even visible to them.

The pragmatic solution is i) to confine biodiversity indicators to taxonomic groups or topics that are relatively well known, i.e. plants and vertebrates; and ii) to make use of qualitative (including informal) as well as quantitative information sources, at least until some parity in knowledge is achieved. In well-studied ecosystem types, biodiversity loss can be measured based on all well-known taxonomic groups, provided that they are included in an unbiased fashion.

The unwillingness of much of the traditional scientific community in the biodiversity domain to be pragmatic rather than perfectly rigorous is a significant impediment to addressing the urgent problem of biodiversity loss. Most indicators rely on very similar sets of input information, so the lack of consensus regarding the form of the indicator should not be an excuse to retard the collection and refinement of fundamental, spatially and temporally explicit data on:

- Land cover and use, and the spatial pattern of marine resource use
- The distribution of species, especially of the plants and vertebrates
- Trends in population size of key species
- The genetic diversity within domesticated species
- The impact of different land use activities on different species.

A lesson in pragmatism can be drawn from the UN Climate Change Convention. When the Intergovernmental Panel on Climate Change was charged with developing performance indicators for greenhouse gas emissions, which can potentially come from hundreds of activities and tens of thousands of individual sources, it chose 'activity-based approaches' rather than full, source-by-source accounting. This approach establishes a statistical relationship between the intensity of an activity (for instance agriculture) and its impact (in this case, on biodiversity). It calculates the score for a given geographical region by multiplying the area exposed to each activity by the impact factor. The Natural Capital Index and Biodiversity Intactness Index (described below) work in this way.

The increasing ease, and decreasing cost, with which genetic information can be collected may in future alter the emphasis currently placed on species-level information, especially for large groups of organisms whose taxonomy is poorly developed. For instance, in calculating endemism scores, it may be less important how many closely related species there are, than the total genetic diversity in the system. Within domesticated organisms, subject to intense breeding, the species concept is somewhat inapplicable in any case, and genetic methods are already widely applied. At present it is not practical to get full genetic profiles for all organisms. Community profiling is a method of choice for groups such as soil microbes, where bulk sampling is relatively straightforward.

2.6 Biodiversity indicators for policy purposes

Several partially-overlapping lists of desirable attributes of biodiversity indicators for policy purposes exist (CBD 2003a; CBD 2003c; CBD 2003d). An integrated list of criteria would include:

- Relevant to biodiversity policy-making
- Simple and easily understood
- Broadly accepted
- Scientifically credible
- Quantitative
- Normative (allowing comparison with a baseline situation and a policy target)
- Measurable in a sufficiently accurate way at an affordable cost
- Responsive to changes at policy-relevant time and space scales
- Useable for scenarios of future projections
- Allow aggregation and disaggregation between ecosystem, national, and international scales
- Useable in various composite indicators and for different purposes.

To date, the bulk of the biodiversity research and media attention has been on species composition, whereas much of the policy-level justification for biodiversity conservation rests implicitly on ecosystem-level, functional attributes. This results in a mismatch between the type of information available, and that needed for policy. While all stakeholders acknowledge the variety of scales and aspects of biodiversity, in practice biodiversity indicators have focused on the organism scale, and on a single compositional measure, species richness. Although the 'ecosystem approach' is widely espoused, in reality, measures of ecosystem diversity have seldom gone beyond statements about the areal extent of prominent ecosystem types, such as forests.

There is no single, all-purpose, universally 'good' indicator in the field of biodiversity. The challenge is to find a *small* set of complementary indicators (since it is apparent that one indicator will not suffice) that is simultaneously easy to grasp, widely applicable, and sensitive. The suitability of a particular indicator depends on the purpose for which it is used. Even for a specified purpose, there are generally two or three indicators that could satisfy the criteria equally well. On the other hand, once the purpose has been defined, it is possible to eliminate entire categories of indicator as inappropriate. Thereafter, considerations of practicality may further reduce the suitable candidate indicators to two or three options. Some indicators, such as hybrid indicators that are arbitrarily-weighted summations of mixtures of states, pressures and responses, should be avoided.

Various types of biodiversity indices are applicable at different stages in the policy process. Indices that identify priority areas for conservation action, such as 'The Last of the Wild', 'Hotspots' or complementarity indices, are aimed at the planning phase, and are usually calculated once only. Performance monitoring tools, repeated on a regular basis, are operational phase indicators used as early warning measures and for evaluating current and future policies.

The authors of this review have been involved in the development of two closely-related biodiversity indices, which reflect our convictions on what type of measures are likely to meet the criteria listed above, and thereby fulfil policy needs while remaining scientifically legitimate. They are the Natural Capital Index (CBD 1997; Ten Brink *et al.* 2002; CBD 2003c) and the Biodiversity Intactness Index (Scholes & Biggs 2004; Scholes and Biggs 2005). In principle, both measure the deviation of abundance of a broad spectrum of species from some reference state. The NCI uses actual population estimates of a restricted number of species, where necessary supplemented by statistical population models. Its disadvantage is that such comprehensive data are only available in well-studied, relatively low biodiversity areas. The BII is more applicable in species-rich but relatively data-poor regions. It uses a panel of experts to assess the impacts of various types of human actions on abundance within functional groups of species. Its disadvantage is the uncertainty associated with such judgements. The numbers produced by both indices are relatively easy to understand and explain, and they integrate both organism-level information (richness, abundance) and ecosystem-level information (area extent of ecosystems, overlaid by area extent of human activities). They are structural/compositional, but can be adapted for functional views as well, by applying them to functional types.

2.7 Gaps in knowledge and research needs

This chapter has attempted to assess biodiversity concepts and indicators relevant to monitoring progress in sustainable development at national to global scales. While certain aspects of biodiversity have been well researched, key gaps remain in the information needed for policy purposes. These include better information on:

- Functional relationships between biodiversity and ecosystem services, and especially the presence of thresholds where these exist;
- Genetic relatedness and redundancy within and between species;
- Robust predictors, for all major ecosystem types in the differing parts of the world, of the consequences of major human activities, such as extensive and intensive agriculture, harvesting, settlement and industrial pollution, on various categories of biodiversity;
- Consistent and reliable maps of land use (and ocean use) and species distributions at regional and global scales;
- Historical ecology in order to understand processes and construct baselines;
- Biodiversity observation and assessment systems for supplying consistent, long-term data for indicator construction.

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Chapter 3: A biodiversity intactness index

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This chapter introduces the BII as a synthetic, policy-relevant indicator, specifically with regard to assessing progress towards the international target of reducing the rate of biodiversity loss by 2010. The paper establishes the algorithmic basis for the BII and details the method for calculating the index. To illustrate its application, the index is applied to southern Africa (South Africa, Lesotho, Swaziland, Namibia, Botswana, Zimbabwe and Mozambique). The rate of biodiversity loss is estimated for the decade of the 1990s, and the scalability and sensitivity of the index are tested.

The paper was submitted to *Nature* in April 2004. Following review, we were invited to resubmit the paper if we could validate the expert estimates and address questions relating to the sensitivity of the index. This was done by conducting an extensive survey of published field studies, and by incorporating summarized results of the work presented in Chapter 4, which had already been presented at a conference. The paper was subsequently resubmitted to *Nature* in November 2004, accepted in December 2004, and published in March 2005.

R.J. Scholes conceived the original idea for the index (hence his lead authorship), and we then jointly developed it further. I derived the mathematical formulation of the index and its partial forms, as well as the method for calculating the confidence intervals. I also established the procedure for calculating the index on either a vector or raster basis, depending on the data that are available. The former is suitable for calculation using spreadsheet software, and the latter using a Geographic Information System. I conducted half the expert interviews, and carried out the literature validation of the expert estimates. I was responsible for compiling the land use map of southern Africa, including the modelling of degraded areas, under the guidance of R.J. Scholes. The authors jointly derived the rates of land use change between 1990 and 2000. I carried out all the BII calculations, the sensitivity analyses and the test of scalability. I was responsible for coordinating the development and submission of the paper, which was jointly written and revised by the authors.

Chapter 3: A biodiversity intactness index

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The nations of the world have set themselves a target of reducing the rate of biodiversity loss by 2010. We propose an index for assessing progress towards this target that is simple and practical, yet sensitive to important factors that influence biodiversity status, and which satisfies the criteria for policy relevance set by the Convention on Biological Diversity. Application of the Biodiversity Intactness Index (BII) is demonstrated on a 4-million km^2 region of southern Africa. The BII score in the year 2000 is 84 ± 7%: in other words, averaged across all plant and vertebrate species in the region, populations have declined by 16% relative to their presumed pre-modern levels. The rate of loss for the decade of the 1990s is estimated as 0.8% (absolute). The taxonomic group with the greatest loss is mammals at 71% remaining, and the ecosystem type with the greatest loss is grassland at 74% remaining.

3.1 Introduction

The loss of biodiversity in the modern era, at rates unequalled since the major extinction events in the distant geological past (WCMC 2000; Anderson 2001), is a matter of considerable policy concern. The Convention on Biological Diversity (CBD) has adopted a target of reducing the rate of biodiversity loss by 2010 (UN 2002). For this target to be met, a method of measuring biodiversity status must be agreed on and implemented. At present no scientific consensus measure exists (Royal Society 2003), although several candidates have been proposed (Reid *et al.* 1993). The CBD has agreed on a partial set of indicators (CBD 2003b). Difficulties in establishing operational indicators stem largely from the complex, multi-dimensional nature of biodiversity, which can be defined in terms of composition, structure and function at multiple scales (Noss 1990).

The CBD developed a set of criteria that an indicator of biodiversity change should satisfy (CBD 2003a). The salient ones are that it should be scientifically sound, sensitive to changes at policy-relevant spatial and temporal scales, allow for comparison with a baseline situation and policy target, useable in scenarios of future projections, and amenable to aggregation and disaggregation at ecosystem, national and international levels. It should also be simple and easily understood, broadly accepted, and measurable with sufficient accuracy at affordable cost.

While there is no shortage of ways to express biodiversity (CBD 2003a; Magurran 2004), none adequately meet the criteria set by the CBD. Most indices require essentially complete knowledge of the biota or the population sizes of individual species, neither of which are achievable conditions at regional to global scales for the next several decades. Many methods are scale-dependent and thus hard to interpret in a comparative context. The most widely-used indicators are based either on risk of extinction (Hilton-Taylor 2000), or on land area under conservation protection (IUCN & UNEP 2003). Several indices have been offered that combine either a sparse and selective set of population estimates for indicator species (Loh 2002), or combine a number of factors that are thought to relate to biodiversity status (Sanderson *et al.* 2002).

A sensitive, realistic and useful measure of biodiversity loss needs to be based on changes in population abundance, across a wide range of the species, and consider the entire landscape. At a global scale, habitat loss, including reductions in quality and quantity of suitable environment, is the major factor responsible for declines in species abundance (WCMC 2000; Jenkins 2003). Other important causes, such as excessive harvest pressure or the effects of pollutants, can also be expressed on the basis of area affected and intensity of impact. This paper introduces an index that we believe takes account of these factors and meets the CBD criteria for policy relevance.

3.2 Attributes of the proposed index

The Biodiversity Intactness Index (BII) is an indicator of the average abundance of a large and diverse set of organisms in a given geographical area, relative to their reference populations. We recommend calculating BII across *all* species within the broad taxonomic groups that are reasonably well described. For most parts of the world this includes plants and vertebrates, and excludes invertebrates and microbes, which are diverse, but poorly documented. We exclude alien species in the calculation of the index.

An easily grasped reference population for large parts of the world is that which occurred in the landscape before alteration by modern industrial society. Since accurate data on pre-modern populations are seldom available, contemporary populations in large protected areas serve as a practical reference. Alternative reference points can be defined for parts of the world already highly transformed by the beginning of the modern period, for example, by selecting a specific baseline year within record or reliable memory.

The BII can in principle be calculated exactly by 'bottom-up' aggregation of population data for individual species. However, especially in the highly biodiverse, but poorly-studied parts of the world, this will not be a practical option for the next several decades. The proposed strategy is therefore to initially calculate the BII 'top-down'. This is analogous to the top-down/bottom-up distinction that has permitted progress in the assessment of global greenhouse gas emissions (Houghton *et al.* 1997). In the greenhouse gas example, the collective national emissions from thousands of individual sources are estimated on an activity basis, rather than source-by-source summation. In the case of biodiversity, we estimate the impacts of a set of land use activities on the population sizes of groups of ecologically similar species ('functional types'). The chosen land use activities range from complete protection to extreme transformation, such as urbanisation. All activities are expressed on the basis of area affected. The index is aggregated by weighting by the area subject to each activity and the number of species occurring in the particular area.

The BII is an aggregate index, intended to provide an intuitive, high-level synthetic overview for the public and policy-makers. It can be disaggregated in several ways to meet the information needs of particular users: by ecosystem or political units, taxonomic group, functional type, or land use activity. This provides transparency and credibility. The BII has the same meaning at all spatial scales. It is possible to estimate the value of BII for the past, and project it into the future under various scenarios. An error bar can be associated with the BII, allowing a monitoring goal to be defined in terms of shrinking the uncertainty range.

3.3 The Biodiversity Intactness Index algorithm

The BII gives the average richness- and area-weighted impact of a set of activities on the populations of a given group of organisms in a specific area. The population impact (I_{ijk}) is defined as the population of species group *i* under land use activity *k* in ecosystem *j*, relative to a reference population in the same ecosystem type. The BII is calculated as:

$$BII = \frac{\sum_i \sum_j \sum_k \mathrm{R_{ij}A_{jk}I_{ijk}}}{\sum_i \sum_j \sum_k \mathrm{R_{ij}A_{jk}}}$$

Where

R_{ij} = Richness (number of species) of taxon *i* in ecosystem *j*

A_{jk} = Area of land use *k* in ecosystem *j*

The BII can be disaggregated along different axes. For instance, the intactness of a particular taxonomic group *i* is given by

$$BII_i = \frac{\sum_{\mathrm{j}} \sum_k \mathrm{R_{ij}A_{jk}I_{ijk}}}{\sum_{\mathrm{j}} \sum_k \mathrm{R_{ij}A_{jk}}}$$

For communication to the public, we suggest expressing BII as a percentage rather than a proportion.

Species-by-species population data for estimating I_{ijk} is seldom available for more than a few species, in a few locations. We used expert judgement to generate a matrix of values of I_{ijk} for southern Africa. Three or more highly experienced specialists in each broad taxonomic group (plants, mammals, birds, reptiles and amphibians) were independently asked to estimate the reduction in the populations of their speciality group caused by a predefined set of land use activities (Table 3.1). Their estimates were made relative to populations in a large protected area in the same ecosystem type, of which six were defined: forest, savanna, grassland, shrubland, *fynbos* (a South African sclerophyllous thicket) and wetland. To assist in the estimation process, each taxonomic group was divided into 5-10 functional types that respond in similar ways to human activities. Across all groups, the functional types were defined primarily by body size, trophic niche and reproductive strategy. I_{ijk} typically assumed values between 0% and 100%, but exceeded 100% in situations where certain activities benefited particular functional types. Estimates of I_{ijk} were aggregated up to the broad taxonomic level by weighting the estimates for each functional type by the number of species in that group in the particular ecosystem type.

Table 3.1 Classes and data sources used to compile a land use map.

Land use class	Description	Examples	Data source
Protected	Minimal recent human impact on structure, composition or function of the ecosystem. Biotic populations inferred to be near their potential.	Large protected areas, national, provincial and private nature reserves, 'wilderness' areas.	World Database on Protected Areas (IUCN & UNEP 2003). All designated protected areas of IUCN categories I-V.
Moderate use	Extractive use of populations and associated disturbance, but not enough to cause continuing or irreversible declines in populations. Processes, communities and populations largely intact.	Forest areas used by indigenous peoples or under sustainable, low impact forestry; grasslands grazed within their sustainable carrying capacity.	All remaining areas not classified into one of the other five categories.
Degraded	Extractive use at a rate exceeding replenishment and widespread disturbance. Often associated with high human population densities and poverty in rural areas. Productive capacity reduced to approximately 60% of 'natural' state.	Clear-cut logging, areas subject to intense harvesting, hunting, fishing or overgrazing, areas invaded by alien vegetation.	All areas falling below 75% (forest, grassland and savanna) or 50% (shrublands) of expected production as estimated by non-linear regression (Michaelis-Menten function) of maximum annual NDVI on growth days. Degraded areas not estimated for desert, wetland and *fynbos*.
Cultivated	Natural land cover replaced by planted crops. Most processes persist, but are significantly disrupted by ploughing and harvesting activities. Residual biodiversity persists in the landscape, mainly in set-asides and in strips between fields (matrix), assumed to constitute approximately 20% of class.	Commercial and subsistence crop agriculture, both irrigated and dryland, including planted pastures and fallow or recently abandoned cultivated areas. Orchards and vineyards.	SADC Landcover Dataset (CSIR 2002), filled with GLC2000 (EC 2003) for Namibia and Botswana.
Plantation	Natural land cover permanently replaced by dense plantations of trees. Unplanted areas assumed to constitute approximately 25% of class.	Plantation forestry, typically *Pinus* and *Eucalyptus* species.	SADC Landcover Dataset (CSIR 2002)
Urban	Land cover replaced by hard surfaces such as roads and buildings. Dense populations of people. Most ecological processes are highly modified. Remnant semi-natural cover assumed to constitute 10% of class.	Dense human settlements, industrial areas, transport infrastructure, mines and quarries.	Urban extents (CIESIN *et al.* 2004)

A total of 4650 estimates of I_{ijk} were made, comprising five broad taxonomic groups, each with at least three experts, six ecosystem types, an average of six land use activities and eight functional types. This works out at approximately 300 estimates per expert, a process that took about five hours per interview. Estimating the BII for southern Africa therefore took a few weeks of effort, rather than the decades needed for detailed population surveys. The range of estimates from different experts was used to construct an uncertainty bar around I_{ijk} (Figure 3.1). Expert-derived estimates of I_{ijk} were validated against measurements available in the literature (Richardson and van Wilgen 1986; Armstrong and van Hensbergen 1994; Parsons *et al.* 1997; Herremans 1998; Joubert and Ryan 1999; Shackleton 2000; Johnson *et al.* 2002; Meik *et al.* 2002; Dean *et al.* 2002; Fabricius *et al.* 2003; Smart *et al.* 2005) (Figure 3.2). The paucity of field studies, and the fact that the variation between comparable field studies is substantially larger than between the expert estimates, supports the use of expert-based approaches at this time.

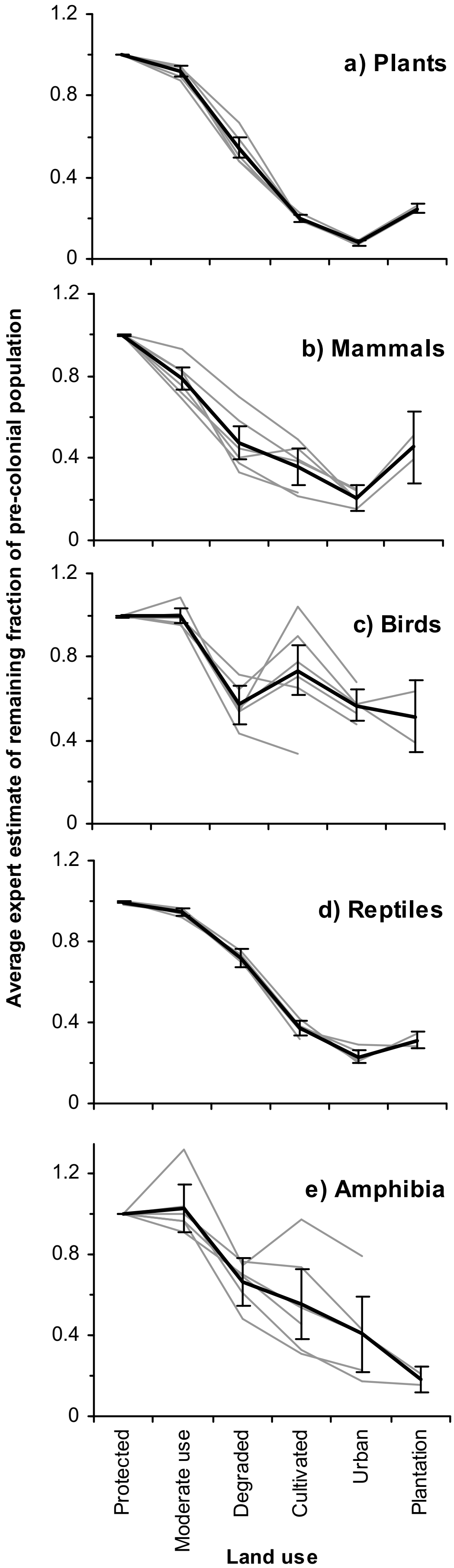

Figure 3.1 Average fraction of original populations remaining under a range of land use activities, as estimated by three or more taxon experts for each broad taxonomic group (a-e). The grey lines reflect the average estimates for each of six biomes or ecosystem types, while the bold line reflects the estimated impact averaged across all biomes. The error bar reflects the 95% confidence interval around the mean estimate.

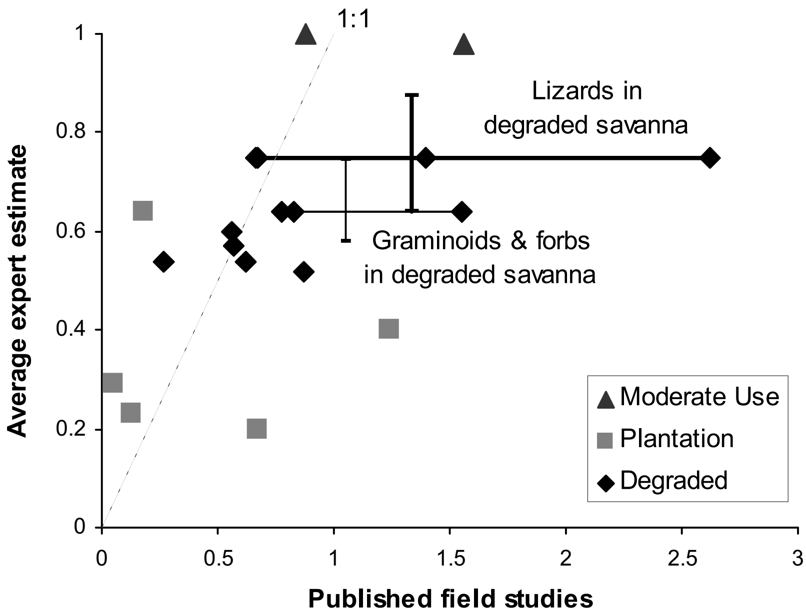

Figure 3.2 Expert estimates I_{ijk} of the ratio of current populations to reference populations under various land use activities, related to field studies published in the literature (n=19, r=0.47, p=0.04). For the two functional groups (lizards and graminoids & forbs) where several comparable field studies were available, the range reported by field studies was substantially larger than the range of estimates between experts.

Species richness data (R_{ij}) is typically available as total species counts per ecosystem type. In this study, species richness data (WWF-US 2003) associated with the WWF Ecoregions (Olson *et al.* 2001) were used. Using such data is equivalent to assuming that every species occurs throughout the extent of the particular ecosystem type. The BII can also be calculated using the potential geographical distributions for individual species, where such data are available.

A_{jk}, the area of a particular land use within a specific ecosystem type, is determined by overlaying land use and ecosystem maps. In this study, broad classes of land use were inferred from land cover and land tenure boundaries. We suggest limiting the number of land use classes to below ten, in order to keep the number of I_{ijk} estimates required manageable. We defined and mapped six levels of land use intensity (Table 3.1). Where classes derived from different data sources overlapped, the highest impact land use was assigned. The resolution of the land use classification affects the estimation of I_{ijk}. In regional-to-global studies, land cover is typically mapped at about 1 x 1 km resolution, and an area classified as, for example, 'cultivated' almost always has inclusions of uncultivated land. Experts were instructed to take this into account in making their estimates of I_{ijk}.

How finely the taxonomic groups need to be divided into functional types, the broad biomes into particular ecosystem types, and how many land use activities are defined is largely a pragmatic question. Our experience is that the patterns of impact of a given land use activity is remarkably similar between ecosystem types (confirmed by ANOVA, which showed no significant biome by land use interaction). This suggests that defining one set of impacts per functional type per biome (high-level ecosystem classification) is sufficient, despite the fact that even the relatively coarse-level ecoregion classification we used for species richness (Olson *et al.* 2001) has several ecosystem types per biome.

The uncertainty in BII can be calculated from the standard errors of each component in the BII equation. We found that by far the largest error was associated with I_{ijk}. We used this error to calculate a 95% confidence interval for BII. The tested the sensitivity of BII to different resolutions and sources of data for A_{jk} and R_{ij} and found it to be small in the southern African case.

3.4 The biodiversity intactness of southern Africa

We applied the BII to the region of southern Africa comprised of South Africa, Namibia, Lesotho, Swaziland, Botswana, Zimbabwe and Mozambique. This region was selected because the experts we interviewed felt comfortable extrapolating their experience within this biogeographical domain, but not beyond.

Overall, we estimate that 84 ± 7% of the pre-colonial number of wild organisms persist in present day southern Africa (Table 3.2), despite greatly increased human demands on ecosystems over the past 300 years. In contrast, over 99% of the species persist (WCMC 2000), illustrating the insensitivity of indices based on extinction (changes in richness alone). Protected areas currently constitute 8% of the region, moderate use areas 78%, degraded areas 2%, cultivated areas 9%, plantations 0.5% and urban areas 2% (Table 3.1). The persistence of populations of wild organisms is therefore substantially greater than that indicated by the area under formal conservation, and points to the importance of areas outside of reserves in the preservation of biodiversity. The contemporary distribution of wild organisms in southern Africa is 85% in moderate use areas, 10% in protected areas, and the remaining 5% in cultivated, degraded, urban and plantation areas.

Table 3.2 BII (%) for southern Africa, per major ecosystem type and broad taxonomic group.

	Area (km^2)	Plants	Mammals	Birds	Reptiles	Amphibia	ALL TAXA
*Richness**		*23,420*	*258*	*694*	*363*	*111*	***24,846***
Forest	*176 893*	75.5	74.9	92.0	85.7	84.8	**78.0**
Savanna	*2 329 550*	85.5	73.2	95.5	88.9	95.9	**87.0**
Grassland	*408 874*	72.5	55.1	90.0	75.6	81.1	**74.1**
Shrubland	*750 217*	86.0	72.2	105.6	93.4	126.5	**88.6**
Fynbos	*78 533*	75.5	78.1	91.0	76.5	79.4	**76.4**
Wetland**	*95 166*	90.7	83.3	94.3	91.7	94.6	**91.3**
ALL BIOMES	***3 839 233***	**82.4**	**71.3**	**96.0**	**88.1**	**95.1**	**84.4**

*Data for South Africa (Le Roux 2002), presented for indicative purposes only

** Refers only to large wetlands, such as the Okavango Delta.

The impact of humans on biodiversity is expressed very selectively. Large-bodied mammal, bird and reptile species that are easy to hunt or harvest are most highly impacted, especially if they are valuable or in direct conflict with human well-being. These species are only a small portion of the total biodiversity. The vast majority of species are impacted mainly through loss of habitat to cultivation or urban settlement, both of which make up relatively small fractions of the southern African landscape.

The greatest impact on biodiversity in southern Africa has occurred in the grassland biome, followed closely by *fynbos*. In both cases, the major cause is conversion to cultivated land, followed by urban sprawl and plantation forestry. The arid shrublands, and savannas that make up most of the land area of southern Africa are overall less impacted. The major cause of biodiversity loss in these systems is land degradation, defined here as land uses that do not alter the cover type, but lead to a persistent loss in ecosystem productivity. Aggregated at a national level, the most densely populated countries in the region, Lesotho and Swaziland, have lower BII scores than the sparsely-inhabited countries of Botswana and Namibia (Table 3.3).

Table 3.3 BII calculated per country for southern Africa.

Country	BII (%)	95% CI*
Botswana	88.5	±7.2
Lesotho	69.0	±7.1
Mozambique	89.5	±8.4
Namibia	91.4	±7.1
Swaziland	72.1	±7.1
South Africa	79.9	±6.5
Zimbabwe	76.2	±7.7
REGION	**84.4**	**±7.3**

* The confidence interval reflects the uncertainty in the expert estimates, I_{ijk}.

We estimated the rate of change of BII in southern Africa during the decade of the 1990s as an absolute decline of 0.8%. We used data on average rates of land cover change in the region for this calculation. Over the period 1991 to 2000, the *relative* increase in protected areas was 3.3% (IUCN & UNEP 2003), cultivated areas 10.1% (FAO 2004), urban areas 5.1% (FAO 2004), and plantation forestry 9.2% (FAO 2001). Degradation was assumed to have kept pace with the growth of the rural population (FAO 2004) (12.9%), and the moderate use category was calculated by difference (-1.9%). Assuming that the I_{ijk} values remained constant over the decade, we calculated the overall BII for the year 1990 as 85.2%.

Our results suggest that the policy action with the greatest potential to prevent further loss of biodiversity in southern Africa is to prevent the extensive areas currently under moderate extractive use from becoming degraded. Moderately used land (e.g. grazed within stocking norms) has almost the same level of biodiversity as protected areas. Degradation, in the form of overgrazing, forest clearing or dense invasion by alien plants, on average reduces species populations by 40-60%. Cultivation and urbanisation have a higher impact per unit land area than degradation, but a much smaller fraction of southern Africa is at risk.

3.5 Scalability and sensitivity of the index

A central feature of the BII is that it can be compared directly within and across scales. This was tested in practice by applying the BII to the three levels of environmental decision-making in South Africa: national (1.2 million km^2), provincial (average area 135 thousand km^2) and local government (average area 4.6 thousand km^2), using biome-level species richness data compiled for South Africa by Le Roux (2002) in place of the WWF ecoregion richness data (WWF-US 2003). The BII was found to deliver intuitively meaningful results at all scales.

Sensitivity to the resolution of species richness data was examined by comparing the values of BII for mammals obtained using individual mammal species distribution maps (Friedmann & Daly 2004), to that obtained using aggregated biome-level mammal richness data (Le Roux 2002). For South Africa overall, the BII for mammals was 67.5% using the biome-level data and 66.1% using the individual species data. At the level of individual biomes, the largest absolute difference between the two methods was 0.6%; the largest difference at both the provincial and municipal levels was 3.0%. No correlation was found between the calculated difference and the size of the units of aggregation.

Sensitivity to the type of species richness data was further explored by calculating 'functional biodiversity intactness' (as opposed to compositional biodiversity intactness (Noss 1990)), in which the aggregated taxon-level I_{ijk} estimates and R_{ij} were derived by weighting by functional group, instead of by species richness. Conceptually, this assigns greater weight to heavily impacted, but relatively species-poor groups, such as megaherbivores, and negates the overwhelming effect of plants on the BII. The functional intactness (FII) of South Africa is 81.0%, and therefore slightly higher than the species-weighted BII (79.6%). Mammals showed by far the largest change (-9.7%) in comparison to the species-based estimate, reflecting the high impact of human activities on a few relatively species-poor functional groups. All other taxa showed small changes. These results suggest that in regions lacking species-level richness data, FII could serve as a first approximation for BII: I_{ijk} would be estimated based on a sample of species per functional group, and R_{ij} would be a count of the number of functional groups present in a particular ecosystem type.

Sensitivity of the BII to the source of land use information was explored in two ways. Increasing or decreasing the total area of any single land use category by 5% of its current area, resulted in an absolute change of less than 0.3% in the value of BII. The maximum compounded change across all categories was 0.7%. Secondly, the BII results based on the merged regional land use map (Table 3.1) were compared to those derived from the South African national land cover (CSIR 1996) and protected areas databases (CSIR 2003). The BII score derived from the national-level data is 81.2%, compared to a value of 79.6% using the regional land use map.

In both the case of species richness (R_{ij}) and land use (A_{jk}), the use of reasonable alternative data sources therefore resulted in a difference in the estimated BII (1.4% absolute for species richness and 1.6% for land use) of about twice the magnitude of the estimated decadal change in BII (0.8%), but five times smaller than the uncertainty associated with the impact factors I_{ijk} (6.4%). Thus, for purposes of documenting change, a single type of information source for species richness and land cover should be used for the beginning and end of the period, and the impact factors should be held constant.

3.6 Conclusion

The BII is an indicator of the overall state of biodiversity in a given area, synthesising land use, ecosystem extent, species richness and population abundance data. It is sensitive to the drivers and changes in the populations of species that typify the process of biodiversity loss, and robust to typical variations in data quality.

It is clear that a single index of biodiversity is not sufficient for all purposes. The BII is not intended to highlight individual species that are under threat, and should be used together with indicators such as the IUCN Red List of Threatened Species. Conceptually, the BII is very similar to the Natural Capital Index (NCI) which has been implemented in the Netherlands (Ten Brink *et al.* 2002). However, the method for estimating BII does not require actual population data, and it can therefore complement the NCI in data-sparse regions.

The principal disadvantage of the BII is that it may be insensitive to slow acting, diffuse impacts on biodiversity, for instance the long-term effects of habitat fragmentation, climate change or pollution. However, an indicator does not have to be flawless to be useful. The Gross Domestic Product is a good example: it is a universal standard for comparison, despite its many, well-known problems. Environmental policy could benefit from a 'macro-ecological' indicator of a similar level of generality.

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Chapter 4: Assessing biodiversity intactness at multiple scales

R. Biggs, R.J. Scholes and B. Reyers

Paper presented at the *Bridging Scales and Epistemologies Conference*
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This chapter presents full details of the tests done to verify the application of the BII at multiple policy-relevant scales, and to test its sensitivity to input data of varying quality. It was presented as an oral paper at the *Bridging Scales and Epistemologies: Linking Local knowledge with Global Science in Multi-scale Assessments Conference*, arranged under the auspices of the Millennium Ecosystem Assessment (MA). Global environmental change is increasingly understood to have causes and effects that span multiple scales, from the local to the global. Until recently, international scientific assessments have typically focused on a single scale, often the global. In addition, they have been based on a particular Western epistemology (way of knowing), which often excludes local knowledge and ignores cultural values. The MA is the first major international environmental assessment that was specifically designed as a multi-scale effort. It aimed to pay particular attention to sub-global and cross-scale processes and interactions, as well as to accommodate and value different knowledge sources. The conference was arranged to explore some of the theoretical and methodological challenges that have been encountered in conducting integrated environmental assessments.

The paper was co-authored by myself, R.J. Scholes and B. Reyers. I am the first author in recognition of the fact that I was responsible for the carrying out the analyses, took the lead in writing the paper, and presented it at the conference. Using South Africa as a test case, the paper explored the application of the BII at local, provincial and national scales, and tested its sensitivity to variations in the quality and resolution of species richness and land use data. Key results from this paper were subsequently included in the BII algorithm paper (Chapter 3) during the revision of that paper. R.J. Scholes provided general guidance on the analyses, and input on the paper. B. Reyers carried out the summation of the individual mammal distribution grids and provided comments on the paper. The paper has been published online in the electronic proceedings of the conference[1] and on CD.

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[1] www.millenniumassessment.org/en/about.meetings.bridging.proceedings.aspx

Chapter 4: Assessing biodiversity intactness at multiple scales

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The Southern Africa Millennium Ecosystem Assessment developed an index for quantifying the state of biodiversity. The Biodiversity Intactness Index (BII) operates at the species level, and is based on changes in species abundance, rather than changes in richness brought about by extinction. The BII uses spatial data on species richness and land use activities per ecosystem type to weight estimates, provided by taxon experts, of the reduction in abundance of all well-known species under a range of land uses. The result is a single score for a particular spatial area, with a confidence range. The index can be disaggregated in various ways, including spatially and by taxonomic group. This paper tests and confirms the ability to calculate and interpret the index at a variety of policy-relevant scales. It furthermore tests the sensitivity of the BII to normal variations in data quality and resolution of species richness and land use data, and confirms the robustness of the index in these respects.

4.1 Introduction

The loss of biodiversity in the modern era, at rates that appear unequalled since the major extinction events in the distant geological past (WCMC 2000; Anderson 2001), is a matter of considerable policy concern. The World Summit on Sustainable Development in its Johannesburg Plan of Implementation, aims to reduce the rate of biodiversity loss by 2010 (UN 2002). For this and other policy targets to be met, a method of measuring biodiversity status must be agreed on. At present no such consensus measure exists (Royal Society 2003), although several candidates have been proposed (Reid *et al.* 1993; CBD 2003b). Difficulties in establishing operational indicators stem largely from the complex, multi-dimensional nature of biodiversity, which is defined in terms of composition, structure and function at multiple scales (Noss 1990).

While there is no shortage of ways to express biodiversity (CBD 2003a; Magurran 2004), most methods impose unattainable data needs, focus on a single scale and aspect of the biodiversity hierarchy, or are scale-dependent and thus hard to interpret in a comparative context. For instance, the most widely used biodiversity metric, species richness, is more sensitive to new discoveries resulting from increased inventory effort than it is to species loss resulting from extinction. Secondly, the relationship between species richness and area is non-linear and ecosystem-type dependent: what is a high species richness number for a small area is a low species richness in a large area.

The Biodiversity Intactness Index (BII) developed for use in the Southern African Millennium Ecosystem Assessment addresses both the problems of excessive data requirements and scale dependence. The BII is an indicator of the state of biological diversity within a given geographical area, which may coincide with a political or ecological boundary, or any other defined region. BII is formally defined as the average, across all species chosen for consideration (typically, the well-known taxonomic groups: plants, mammals, birds, reptiles and amphibians), of the change in population size relative to a reference population. The reference population is conceived as that occurring in the landscape before it was altered by modern industrial society during the colonial period (i.e. pre-1700). In the southern African context, the current populations in large protected areas in each ecosystem type serve as a proxy for the reference populations.

The BII is an aggregate index, intended to provide an intuitive, high-level synthetic overview for the public and policy makers. At the same time, it can be disaggregated spatially, or by taxonomic group, to meet the information needs of various users, providing transparency and credibility. Mathematically and conceptually the BII has the same meaning at all spatial scales, and can thus be compared within and across scales, as well as between geographically separated areas of the same or different sizes. It is possible to estimate the value of BII for the past, and project it into the future under various scenarios of land use change.

The data available for computing BII and other indices vary in quality and resolution, and this variability is itself scale dependent. At small scales (for instance, a single well-studied protected area) very complete and high-resolution data may be available, whereas at the scales relevant to national or international policy, data is typically patchy and of coarse resolution. The BII is able to use data of differing quality and levels of detail as input, making it applicable immediately, but amenable to incremental improvement in the future. An error bar can be associated with the BII, and the long-term research and observation goal can then be defined in terms of shrinking this uncertainty range.

The objectives of this paper are two-fold. Firstly, to illustrate that the index can be meaningfully scaled up and down, and therefore usefully and consistently applied at the typical levels of environmental decision-making. In the case of South Africa, these are the constitutionally-defined three tiers of government: national (1.2 million km^2), provincial (9 units, average area 135461 km^2) and local (262 units, average area 4647 km^2). The second objective is to test the sensitivity of the index to the resolution and quality of the input data used. In particular, we aim to explore the sensitivity of the BII to improved spatial estimates of species richness and to a higher-resolution land use map.

4.2 The Biodiversity Intactness Index algorithm

The derivation and logic of the BII are given in Scholes and Biggs (*in press*). In essence, BII is a richness-and-area weighted average of the population impact of a set of land use activities, on a given groups of organisms, in a given area. If the population impact (I_{ijk}) is defined as the relative population of taxon i (as compared to the reference state) under land use activity k in ecosystem j, then BII gives the average remaining fraction of the populations of all species considered:

$$BII = \frac{\sum_i \sum_j \sum_k R_{ij} A_{jk} I_{ijk}}{\sum_i \sum_j \sum_k R_{ij} A_{jk}}$$

where

R_{ij} = Richness (number of species) of taxon i in ecosystem j

A_{jk} = Area of land use k in ecosystem j

BII can be disaggregated to successive levels of detail along different axes. For instance, the intactness of a particular taxonomic group i, and for a particular taxon i in a given ecosystem j are respectively given by:

$$\mathrm{BII}_i = \frac{\sum_j \sum_k R_{ij} A_{jk} I_{ijk}}{\sum_j \sum_k R_{ij} A_{jk}} \qquad \mathrm{BII}_{ij} = \frac{\sum_k A_{jk} I_{ijk}}{\sum_k A_{jk}}$$

Note that I_{ijk} typically lies between 0 and 1, but can take on values greater than one in some circumstances. For example, cultivation greatly increases the populations of certain categories of bird species, such as granivores, relative to 'natural' areas. Similarly, frugivores are enhanced in urban areas.

4.3 Data sources

I_{ijk} is a matrix of estimates of the fraction of the original populations of specific taxa that persist under a given land use activity in a particular ecosystem. While I_{ijk} can in principle be exactly measured, the species-by-species population data needed to do so are currently available only for a few species in a few locations. In the study that developed the BII (Scholes and Biggs 2005), expert judgement was used to generate the matrix of values of I_{ijk}. Three or more highly-experienced specialists for each selected taxonomic group (plants, mammals, birds, reptiles and amphibians) independently estimated the degree of reduction of populations, relative to populations in a large protected area in the same ecosystem type, caused by a pre-defined set of land use activities (Table 4.1). This was achieved by dividing each taxonomic group into 5-10 functional types that responded in similar ways to human activities. Aggregation up to the broad taxonomic level (birds, mammals etc.) was done by weighting the I_{ijk} estimates for each functional type by the number of species in that functional type in the particular ecosystem. It is also possible to calculate a 'functional biodiversity intactness index' (FII), in which weighting is done by functional group rather than by species richness. I_{ijk} estimates are conceivably sensitive to the resolution of the land use classification used in the calculation of BII. In this study, a 1 x 1 km resolution was used so that, for example, the 'cultivated' category would generally include a fraction of uncultivated land. Experts were asked to account for this in making their population impact estimates.

Table 4.1 Classes used to derive a land use map from land cover, land tenure boundaries and other available information. Two classifications were used: (1) a classification used by Scholes & Biggs (2005) for the southern African analysis, and (2) a classification largely based on the South African national land-cover map. Classification was carried out at a 1 x 1 km resolution and based on the dominant category within a particular unit. A 'cultivated' unit would therefore consist predominantly of cultivated fields, but usually also contain a fraction of uncultivated land. Where classes overlapped, land use was assigned in the following order of priority: urban, plantation, cultivated, degraded, protected, moderate use.

Land use class	Description	Data source
Protected	Minimal recent human impact on structure, composition or function of the ecosystem. Biotic populations inferred to be near their potential. *Examples: Large protected areas, 'wilderness' areas.*	(1) World Database on Protected Areas (IUCN & UNEP 2003). All designated protected areas of IUCN categories I-V. (2) South African national protected areas (CSIR 2003)
Moderate use	Some extractive use of populations and associated disturbance, but not enough to cause continuing or irreversible declines in populations. Processes, communities and populations largely intact. *Examples: Forest areas used by indigenous peoples or under sustainable, low impact forestry; grasslands grazed within their sustainable carrying capacity.*	All remaining areas not classified into one of the other five categories.
Degraded	High extractive use and widespread disturbance, typically associated with large human populations in rural areas. Productive capacity reduced to approximately 60% of 'natural' state. *Examples: Clear-cut logging, areas subject to intense harvesting, hunting or fishing, areas invaded by alien vegetation.*	(1) All areas falling below 75% (forest, grassland and savanna) or 50% (shrublands) of expected production as estimated by non-linear regression (Michaelis-Menten function) of maximum annual NDVI on growth days. Degraded areas not estimated for desert, wetland and *fynbos*. (2) South African national land-cover map (CSIR 1996)
Cultivated	Land cover permanently replaced by planted crops. Most processes persist, but are significantly disrupted by ploughing and harvesting activities. Residual biodiversity persists in the landscape, mainly in set-asides and in strips between fields (matrix), assumed to constitute approximately 20% of class. *Examples: Commercial and subsistence crop agriculture.*	(1) SADC Landcover Dataset (CSIR 2002), filled with GLC2000 (EC 2003) for Namibia and Botswana. (2) South African national land-cover map (CSIR 1996)
Plantation	Land cover permanently replaced by timber plantations. Matrix areas assumed to constitute approximately 25% of class. *Examples: Plantation forestry, typically pinus and eucalyptus species.*	(1) SADC Landcover Dataset (CSIR 2002) (2) South African national land-cover map (CSIR 1996)
Urban	Land cover replaced by hard surfaces such as roads and buildings. Dense populations of people. Most processes are highly modified. Matrix assumed to constitute 10% of class. *Examples: Dense urban and industrial areas, mines and quarries.*	1) Urban extents (CIESIN *et al.* 2004) (2) South African national land-cover map (CSIR 1996)

R_{ij} is the species richness per broad taxon (plants, mammals etc), per ecosystem type. Species richness data is typically available as total species counts per ecosystem and the assumption is made that every species occurs throughout the extent of the particular ecosystem type. (Le Roux 2002) recently compiled such data for South Africa, for eight taxonomic groups in seven biomes; these data were used in this study. Individual species distribution grids are available digitally for certain taxonomic groups in South Africa. The BII is based on potential distribution ranges, and in this paper, the potential mammal distribution data of Friedmann and Daly (2004) was used to explore the sensitivity of BII to the input of more highly resolved spatial species richness data. The potential distribution maps of 272 mammal species were rasterized, and the total number of mammal species per 1 x 1 km grid cell calculated as an alternative R_{ij} input for BII.

A_{jk} is the area of a particular land use within a specific ecosystem, derived by overlaying a land use map on an ecosystem-type map. Ecosystem types were defined as the biomes of South Africa, the boundaries of which were obtained from the National Botanical Institute (Rutherford and Westfall 1994; Low & Rebelo 1996). In this study, the assumption was made that broad classes of land use can be inferred from land-cover and land tenure boundaries. We defined six levels of land use intensity and mapped these by combining data from several global to continental scale data sources (Table 4.1). In order to test the sensitivity of BII to the land use classification employed, a second land use map for South Africa was derived from the South African national land-cover map (CSIR 1996; Fairbanks *et al.* 2000), combined with a national protected areas map (CSIR 2003). The South African national land cover map was derived from high-resolution satellite imagery (fundamental resolution ~20 m, but effectively aggregated to about 500 m), whereas the main source for the southern African regional scale land use map was low to medium resolution imagery with a fundamental resolution of about 1 km.

4.4 Scalability of the index

A central feature of the BII is that conceptually it can be applied at different levels of decision-making, and results can be compared directly within and across scales. This was tested in practice by applying the BII to the three levels of environmental decision-making in South Africa: national, regional and local government (Figure 4.1). The results show that BII delivers intuitively meaningful results down to at least the scale of local government (municipal level). The ability to apply the index at multiple levels of decision-making is very useful: aggregation allows the information to be collapsed to an appropriate level of detail, while disaggregation ensures transparency and allows for the definition of finer-level policy goals. BII can similarly be disaggregated by taxa or biome (Table 4.2); mathematically it is possible to disaggregate BII down to any defined spatial area and any group of species.

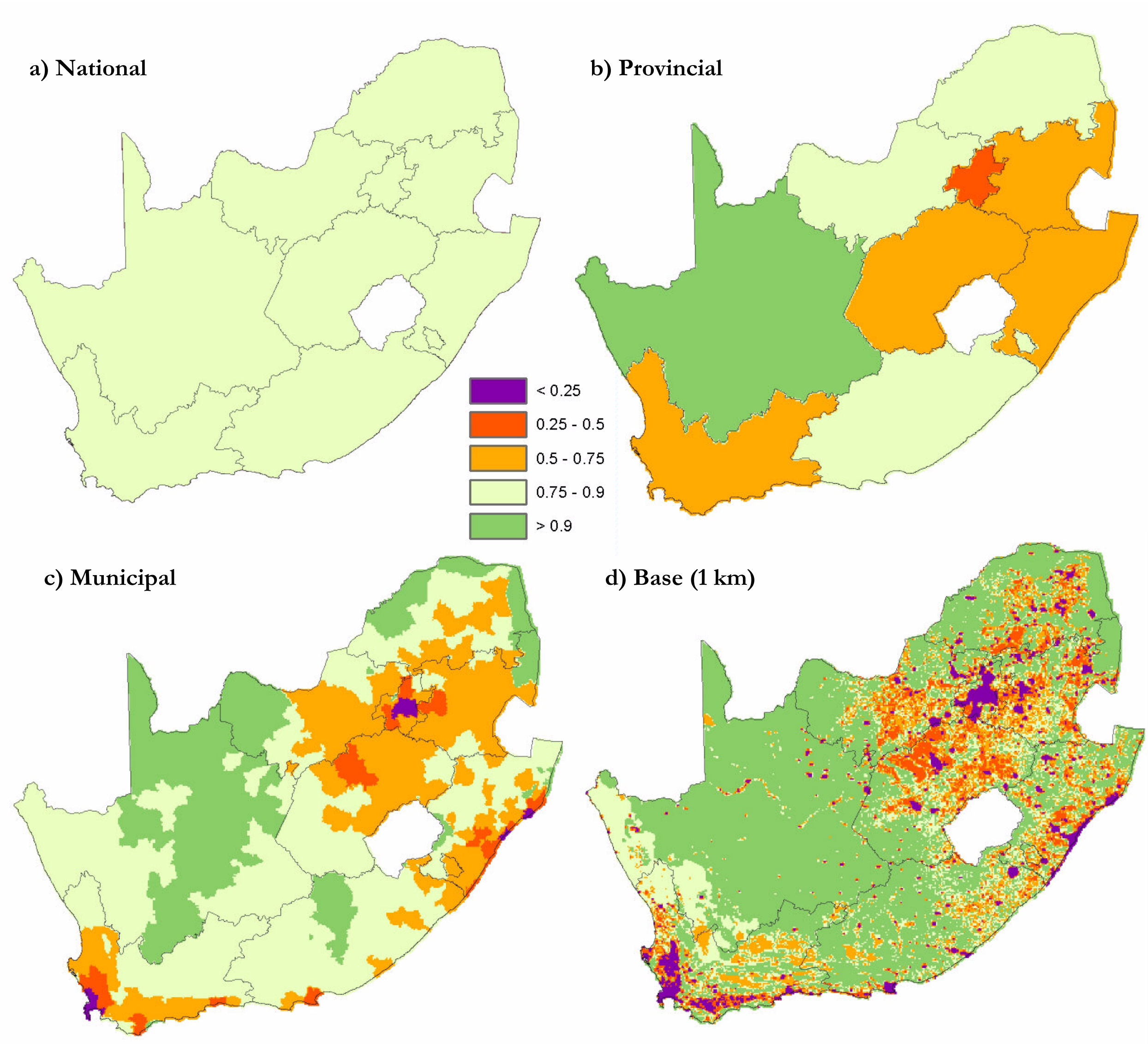

Figure 4.1 The Biodiversity Intactness Index applied at the three levels of environmental decision-making in South Africa: national, provincial and local government. The results are richness and area weighted averages of BII as estimated at a base resolution of 1 km. Values of BII obtained at different scales are directly comparable: they refer to the average abundance of all species in the particular area, expressed as a fraction of pre -industrial era abundance.

Table 4.2 A cross-tabulation of BII per major ecosystem type and broad taxonomic group for South Africa, showing how the overall score (0.80 ± 0.06 at the 95% level) can be disaggregated into column or row scores for ecosystems or taxa, or down to taxa within ecosystems. Note that the aggregations are not simple averages, but are weighted by biome-level species richness (Le Roux 2002) and area (southern African regional scale land use classification given in Table 4.1).

	Area (km^2)	**Plants**	**Mammals**	**Birds**	**Reptiles**	**Amphibia**	**ALL TAXA**
Richness		*23420*	*258*	*694*	*363*	*111*	***24846***
Forest	*7 148*	0.68	0.71	0.84	0.77	0.75	**0.69**
Savanna	*416 484*	0.82	0.71	0.93	0.86	0.93	**0.83**
Thicket	*41 349*	0.78	0.66	0.89	0.82	0.90	**0.78**
Grassland	*294 815*	0.69	0.54	0.90	0.73	0.78	**0.71**
Nama Karoo	*297 810*	0.86	0.70	1.06	0.93	1.29	**0.88**
Succulent Karoo	*82 490*	0.83	0.69	1.03	0.90	1.25	**0.84**
Fynbos	*77 111*	0.70	0.78	0.91	0.77	0.79	**0.71**
ALL BIOMES	***1 217 207***	**0.78**	**0.68**	**0.95**	**0.84**	**0.92**	**0.80**

Functional diversity intactness

The possible use of BII as an indicator of functional biodiversity intactness (as opposed to compositional biodiversity intactness, see (Noss 1990)) was explored by weighting the taxon-level I_{ijk} estimates by functional group, as defined by the experts, instead of by species richness. In principle this is another form of scaling: it coarsens the resolution along the taxon axis rather than along a geographical scale axis. Conceptually, it would assign greater weight to heavily impacted, but relatively species-poor groups, such as megaherbivores.

The functional intactness (FII) of South Africa, calculated across all taxa, is 0.81, and therefore slightly *higher* than the species-weighted BII (Table 4.3). The FII was lower than the BII in savannas, by 0.038; shrubland and *fynbos* both increased by 0.065. Mammals showed by far the largest decline, 0.097, in comparison to the species-based estimate. Reptiles also showed a slight decline; all other taxa showed marginal increases in the value of FII as compared to BII.

While the functional groups used in this calculation were not specifically defined for the purpose of calculating FII, the results suggest that other than in the case of mammals, there do not appear to be specific functional groups that are currently suffering excessively greater impact than other functional groups. The results indicate that under such circumstances, FII serves as a good proxy to BII. This suggests that in regions lacking species-level data, FII could serve as a first approximation for BII: I_{ijk} would be estimated based on a sample of species per functional group, and R_{ij} would be a count of the number of functional groups present in a particular biome.

Table 4.3 Results of a sensitivity analysis of BII to inputs of species richness and land use. *Column b:* Comparison of BII values (column a) to those of FII (Functional Intactness Index), where FII is derived by weighting I_{ijk} by functional group instead of by species richness; *Column c:* Comparative values of species-weighted BII as derived from the regional-scale land use classification (1) and the national-level classification (2) (see Table 4.1); and *Columns d and e:* Comparison of the values of BII obtained for mammals using aggregated biome-level richness data as opposed to individual species distribution grids. Note that the results presented here are for the biomes as defined in the expert interviews: the thicket biome is therefore included with savanna, and the Succulent and Nama Karoo are lumped together as shrubland.

	a	b	c	d	e
				Mammals	
Index	**BII**	**FII**	**BII**	**BII**	**BII**
Land use	**(1)**	**(1)**	**(2)**	**(1)**	**(1)**
Species richness	**Biome**	**Biome**	**Biome**	**Biome**	**Grid**
Biomes					
Forest	0.690	0.734	0.761	0.706	0.712
Savanna	0.826	0.788	0.831	0.708	0.700
Grassland	0.708	0.713	0.731	0.537	0.538
Shrubland	0.866	0.931	0.892	0.699	0.696
Fynbos	0.706	0.770	0.737	0.778	0.772
Taxa					
Plants	0.784	0.832	0.802		
Mammals	0.676	0.579	0.686	**0.675**	**0.661**
Birds	0.949	0.960	0.949		
Reptiles	0.840	0.798	0.857		
Amphibia	0.923	0.924	0.936		
SOUTH AFRICA	**0.796**	**0.810**	**0.812**		

4.5 Sensitivity of the index

Ideally, any biodiversity index should be sufficiently sensitive to reflect the impacts of changing land use over time, but should not be overly sensitive to differences in the interpretation of land use classes or estimates of species richness that are within the normal uncertainty in those parameters.

Sensitivity to R_{ij} (species richness)

Sensitivity to the spatial resolution of species richness data was explored by comparing the values of BII obtained using the richness surface derived by summation of individual mammal distribution grids, as opposed to using biome-level mammal species richness data (i.e. the same richness value is assigned to every grid cell in a particular biome).

For South Africa overall (area 1.2 million km^2), BII for mammals declined from 0.675 using the biome-level data to 0.661 using the individual species distribution data (Table 4.3). At the level of individual biomes, the largest difference between the two methods was 0.006, and at both the provincial and municipal levels the largest difference was 0.03. No correlation was found between the calculated difference and the size of the units of aggregation.

At the scales at which we envisage BII being applied (areas of 500 km^2 upward), it appears robust to the use of coarse-level species richness data, at least in the case of mammals. This may be due to the fact that the distribution ranges of a large fraction of mammal species coincide with biome boundaries (Erasmus *et al.* 2002). In the case of amphibians and reptiles, which are less mobile and more habitat-dependent than mammals, resulting in a larger fraction of the species having very restricted ranges, the robustness of BII to the use of coarse-resolution species richness data must still be verified. Where individual species distribution data are available, the estimation of BII can be further improved by using the disaggregated functional group level estimates of I_{ijk}, rather than the weighted average taxon level estimates of I_{ijk} in the calculation of BII.

Sensitivity to A_{jk} (land use classification)

The results presented thus far are based on the global to continental scale data sources used in Scholes and Biggs (2005) to map land uses for the southern African application of BII (Table 4.1, data source (1)). In this section, a national scale land use classification (Table 4.1, data source (2)) was used to explore the sensitivity of BII to the input A_{jk}, the area in each ecosystem type under a given land use category. Table 4.4 summarizes the relative proportions of South Africa apportioned to the different classes using the two data sources.

Table 4.4 Comparison of the percentage of South Africa classified into the classes described in Table 4.1. Area (1) refers to the land use classification derived from the regional -scale data sources, while Area (2) refers to the classification derived from na tional-scale data (see Table 4.1). There is a good correspondence between the two classifications for all classes except "degraded" and "urban".

Land use	% Area (1)	% Area (2)	Area (1)/ Area (2)
Protected	5.00	6.96	0.72
Moderate use	75.59	74.22	1.02
Degraded	2.34	4.68	0.50
Cultivated	10.68	11.73	0.91
Plantation	1.22	1.26	0.97
Urban	5.17	1.14	4.52

The land use classification derived from the South African national -level data gives a BII value of 0.812, compared to the value of 0.796 (Table 4.3) dete rmined using the southern African regional-scale data. The BII values for all taxa and all biomes were marginally higher (less than 0.03) in the national than regional products, except for the forest biome, which increased by 0.07. While there are large differences in the total area assigned to certain classes (for example the urban and degraded categories), these are generally classes that, while having a high impact, constitute a relatively small fraction of the total land surface. Consequently, differenc es in their classification have a limited impact on the value of BII. In addition, classes where there are large differences in the areas assigned are often classes that are partially overlapping, as evidenced by the fact that the sum of their areas is very similar using the different data sources (e.g. the sum of urban and degraded is 5.8% using national-level data, and 7.5% using regional-level data).

These results suggest that, at the coarse level of land use classification used in this study (six broad classes of impact), there is sufficient agreement between the main land cover and land tenure data sources that the choice of a particular data source will not significantly change the value of BII.

4.6 Conclusions

- The Biodiversity Intactness Index (BII) can be applied at scales at least down to 500 km^2 (i.e. to the level of local government) while retaining its intuitive meaning.

- The values given by the BII at the national scale (~ 1 million km^2) are robust to variations within the normal range in data quality and resolution, specifically to:
    a. Biome-level species richness data versus species-by-species distribution data;
    b. Missing information on species richness, which can to a degree be substituted by functional type richness;
    c. Reasonable differences in the interpretation of land use classes.

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Chapter 5: A biodiversity score for South Africa

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This chapter presents the results of applying the BII at multiple levels of decision-making in South Africa. The paper uses the best-available South African data to assess the state of biodiversity and explore the implications for biodiversity conservation, specifically in terms of the recently enacted biodiversity legislation in the country. It draws on the analyses performed in Chapter 4, but substitutes the regional land-use and species richness datasets with South African data. Hence the differences between the results presented in Chapter 4 and those presented here.

I took in the lead role in developing, writing and submitting the paper, which is recognized by my lead authorship. I recalculated the BII using South African data, as well as calculating the confidence intervals for the BII per taxonomic group, ecosystem and land use. I carried out the analysis of the contribution that each land use activity made to the reduction in BII experienced by 2000, as well as the exact contribution of each land use activity to the remaining BII. The results of these analyses are presented in this paper in full for the first time. My co-authors, B. Reyers and R.J. Scholes, provided comments and suggestions on the draft manuscript.

Chapter 5: A biodiversity score for South Africa

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South Africa's new Biodiversity Act requires the development of a national biodiversity framework to provide for an integrated approach to biodiversity management in the country. The Act also requires the monitoring and regular reporting of the status of the Republic's biodiversity. In this paper, a new approach is applied to assess the state of biodiversity in South Africa and highlight possible implications for biodiversity management. The Biodiversity Intactness Index (BII) gives the average change in population size of all terrestrial plants and vertebrates relative to their pre-colonial abundances. We estimate that, on average, populations have declined by 19 ±7% over the past three centuries in South Africa. Losses are greatest in the grassland, *fynbos* and forest biomes, and mammals are the most impacted taxonomic group. The BII can also be calculated by province and municipality. It is estimated that 80% of the remaining wild organisms in South Africa are in the extensive areas of the country under grazing management. This suggests that the policy action with the greatest potential to limit further loss of biodiversity is to prevent the degradation of these areas, which could potentially halve the biodiversity in the country.

5.1 Introduction

South Africa's recent Biodiversity Act (No. 10 of 2004) (Republic of South Africa 2004) recognises the importance of the country's rich biological resources to the well-being and prosperity of its citizens. The Biodiversity Act is underpinned by the South African Constitution and Bill of Rights which provides all citizens with the right "*to have the environment protected, for the benefit of present and future generations, through reasonable legislative and other measures that i) prevent pollution and environmental degradation; ii) promote conservation; and iii) secure ecologically sustainable development and* *use of natural resources while promoting justifiable economic and social development*' (Republic of South Africa 1996).

A requirement of the Biodiversity Act is the development, within three years, of a national biodiversity framework (Republic of South Africa 2004). This framework must provide for an integrated, co-ordinated approach to biodiversity management by all governmental and non-governmental stakeholders, as well as identify priority areas for conservation action and the establishment of protected areas. This framework will also help fulfil the requirements for developing national strategies, plans or programmes for the conservation and sustainable use of biodiversity, as set by the Convention on Biological Diversity, to which South Africa is a contracting party[*]. The Biodiversity Act further provides for the establishment of the South African National Biodiversity Institute (SANBI), which is *inter alia* tasked with monitoring and reporting on the status of South Africa's biodiversity, and undertaking and promoting research on the sustainable use of indigenous biological resources.

A key activity informing the development of the national biodiversity framework is the National Spatial Biodiversity Assessment (NSBA) (Driver *et al.* 2004). To date, the main conceptual approach applied by the NSBA has been 'systematic conservation planning'. This method is based on quantitative targets that specify how much of the landscape is needed to conserve a representative set of sample species and habitats, as well as ecological processes. Its application has led to the identification of nine broad priority areas for conservation action[†]. The NSBA also quantifies the extent of each ecosystem type that has been transformed by human activities. This approach, however, provides only a partial assessment of the status of biodiversity in the country, and does not differentiate between the impacts of different types of land use.

The authors of this paper have been involved in the development and application of a new index, the Biodiversity Intactness Index (BII), which we believe can complement the information provided by the NSBA. The BII was conceived for use in the Southern African Millennium Ecosystem Assessment (Biggs *et al.* 2004a) to provide a synoptic overview of biodiversity status. The BII is formally defined as the average, across all well-described taxa (i.e. plants, mammals, birds, reptiles and amphibians), of population sizes relative to a reference population (Scholes and Biggs 2005). For southern Africa, the reference populations are conceptually those present

[*] www.biodiv.org

† www.nbi.co.za/biodiversity/planning.htm

before the substantial alteration of the landscape triggered by European settlement (i.e. pre-1700 populations). Since records of plant and wildlife populations from this era are virtually non-existent, the current populations in large protected areas in each ecosystem type serve as a proxy for the 'pre-colonial state'.

The BII is an aggregate index that combines information on ecosystem distribution, species richness and the extent and impact of major land uses on biodiversity. It is intended to provide an easy-to-understand overview of the state of biodiversity for policy makers and the public. The score can be disaggregated spatially (by ecosystem or political boundaries), by taxonomic group or by land use activity, to meet more detailed information needs of different users. The BII is able to use data of differing quality and levels of detail, and can therefore be calculated using existing data, but is amenable to future improvement. The uncertainty in the estimate of the BII can be quantified and used to identify issues requiring further research.

The BII has been applied to southern African to assess the current state of biodiversity and rate of loss (Scholes & Biggs 2004; Scholes and Biggs 2005), and to project future changes in biodiversity under various scenarios of land use change (Simons *et al. in prep.*). For the southern African region overall, there has been an estimated average decline of 16% in the abundance of terrestrial plants and vertebrates since pre-colonial times, and the loss of BII over the decade of the 1990s has been estimated at 0.8% (absolute) (Scholes and Biggs 2005). A scenario analysis for the region suggests a continuing decline in BII for most of the 21st century, with rates of decline likely to accelerate for several decades (Simons *et al. in prep.*). The BII has also been used to reconstruct changes in biodiversity in South Africa over the twentieth century (Biggs and Scholes *in press*). This work indicated that the populations of terrestrial plants and vertebrates in South Africa have declined, at a varying rate of around 1% per decade, over the past century. In contrast to the situation for the southern African region, there is some evidence that the rate of biodiversity loss in South Africa may be declining.

The objectives of this paper are to apply the BII to South Africa, using the best-available South African data, in order to provide insights that may be useful in the development of the national biodiversity framework. It also aims to introduce a tool that could be used to complement existing methods in reporting on the state of South Africa's biodiversity. We apply the BII per ecosystem as well as at the three main levels of environmental decision-making in South Africa, namely the national, provincial and local government levels. We highlight the implications of our findings for biodiversity management in the country, specifically in terms of sustainable natural resource use, priorities for further research, and the need for coordination between different government departments, specifically the Department of Environmental Affairs and Tourism [*], the Department of Agriculture[†], and the Department of Water Affairs and Forestry[‡].

5.2 Methods

The full derivation and logic of the BII are described in Scholes and Biggs (Scholes and Biggs 2005). In essence, BII is a richness-and-area weighted average of the population impact of a set of land use activities, on a given groups of organisms, in a given area. If the population impact (I_{ijk}) is defined as the relative population of taxon i (as compared to the reference state) under land use activity k in ecosystem j, then BII gives the average remaining fraction of the

$$BII = \frac{\sum_i \sum_j \sum_k R_{ij} A_{jk} I_{ijk}}{\sum_i \sum_j \sum_k R_{ij} A_{jk}}$$

populations of all species considered:
where

R_{ij} = Richness (number of species) of taxon i in ecosystem j
A_{jk} = Area of land use k in ecosystem j

For communication purposes, we suggest expressing BII as a percentage rather than a proportion. Partial summations give the BII per taxonomic group, ecosystem type (biome or finer-level ecoregion), land use activity, and combinations of these factors.

While I_{ijk} can in principle be exactly measured, the species-by-species population data needed to do so are currently available only for a few species in a few locations. In the southern African case, expert judgement was used to generate the matrix of values of I_{ijk} and validated against available field studies (Scholes and Biggs 2005). Three or more highly-experienced specialists for each selected taxonomic group (plants, mammals, birds, reptiles and amphibians) independently

[*] www.environment.gov.za
[†] www.nda.agric.za
[‡] www.dwaf.gov.za

estimated the degree of reduction of populations, relative to populations in a large protected area in the same broad ecosystem type or biome, caused by a pre-defined set of land use activities (Table 5.1). This was achieved by dividing each taxonomic group into 5-10 functional types that responded in similar ways to human activities. I_{ijk} typically lies between 0 and 1, but can take on values greater than one in some circumstances. For example, cultivation greatly increases the populations of certain categories of bird species, such as seed-eaters, relative to 'natural' areas. Aggregation up to the broad taxonomic level (birds, mammals etc.) was done by weighting the expert-derived I_{ijk} estimates for each functional type by the number of species in that functional type in the particular ecosystem.

Table 5.1 Land use classes used to calculate the BII for South Africa. The land use map was compiled from the South African national land cover map (CSIR 1996; Fairbanks *et al.* 2000a) and the national protected areas map (CSIR 2003) at a 1 x 1 km resolution.

Land use class	Description	*Examples*
Protected	Minimal recent human impact on structure, composition or function of the ecosystem. Biotic populations inferred to be near their potential.	Large protected areas, national, provincial and private nature reserves, 'wilderness' areas and mountain catchments.
Moderate use	Extractive use of populations and associated disturbance, but not enough to cause continuing or irreversible declines in populations. Processes, communities and populations largely intact.	Grasslands grazed within their sustainable carrying capacity, forest areas used by indigenous peoples.
Degraded	Extractive use at a rate exceeding replenishment and widespread disturbance. Often associated with high human population densities and poverty in rural areas. Productive capacity reduced to approximately 60% of 'natural' state.	Areas subject to intense harvesting, grazing, hunting or fishing, areas invaded by alien vegetation.
Cultivated	Natural land cover replaced by planted crops. Most processes persist, but are significantly disrupted by ploughing and harvesting activities. Residual biodiversity persists in the landscape, mainly in set-asides and in strips between fields (matrix), assumed to constitute approximately 20% of class.	Commercial and subsistence crop agriculture, both irrigated and dryland, including planted pastures and fallow or recently abandoned cultivated lands. Orchards and vineyards.
Plantation	Natural land cover permanently replaced by dense plantations of trees. Unplanted areas assumed to constitute approximately 25% of class.	Plantation forestry, typically *Pinus* and *Eucalyptus* species.
Urban	Natural land cover replaced by hard surfaces such as roads and buildings. Most ecological processes are highly modified. Remnant semi-natural cover assumed to constitute 10% of class.	Dense human settlements, industrial areas, transport infrastructure, mines and quarries.

R_{ij} is the species richness per broad taxon (plants, mammals etc), per ecosystem type. Data recently compiled by Le Roux (Le Roux 2002) for plants and vertebrates in the seven terrestrial biomes that cover South Africa were used in this study. The use of such data makes the assumption that each species occurs throughout the extent of every biome in which it has been recorded. This is an approximation which we have tested and found to introduce an error at the national level of about one-fifth of the uncertainty associated with the estimates of the impacts of various land uses (Scholes and Biggs 2005). The BII can be calculated more accurately by using the potential distribution maps of individual species. These are available for most, but not all, plant and vertebrate species in South Africa. For studies at finer scales, the use of the more resolved datasets is recommended. For instance, at the level of local government, using biome-level mammal richness data rather than individual species distributions changed the BII score by 3% (Scholes and Biggs 2005). The BII can, if necessary, be calculated using a mixture of biome-level and species-level data; for instance, actual distributions for birds and mammals, but biomes for plants.

A_{jk} is the area of a particular land use within a specific ecosystem, derived by overlaying a land use map on an ecosystem type map. Ecosystem types were defined as the biomes of South Africa (Rutherford and Westfall 1994; Low & Rebelo 1996). Land use for South Africa was derived by combining the 1995 South African national land-cover map (CSIR 1996; Fairbanks *et al.* 2000b) with a national protected areas map (CSIR 2003). A base resolution of 1 x 1 km was used to correspond to the resolution at which the expert estimates were made. At this resolution, cells classified as cultivated or urban would generally include a fraction of uncultivated or non-built-up land; experts were asked to take this into account in making their estimates. The error associated with the use of an alternative land use map has also been tested at the national level, and found to be approximately five times smaller than the error associated with the land use impact estimates (Scholes and Biggs 2005).

5.3 Results and discussion

The biodiversity intactness of South Africa in the year 1995 is estimated as 81.2 ± 6.9% (Table 5.2). In other words, we estimate that averaged across all species of terrestrial plants and vertebrates, populations have declined by 18.8% relative to their presumed pre-colonial levels. The persistence of organisms as estimated by the BII is therefore substantially greater than that indicated by the proportion of the country under formal conservation protection (7%) (CSIR 2003), but considerably less than that indicated by extinction-based measures (less than 0.5% of known organisms in South Africa have become extinct in the modern era (IUCN 2002)).

The pattern of threat to biomes revealed by the BII agrees with the findings of the NSBA (Driver *et al.* 2004). The biomes that have lost the most biodiversity are the grassland, *fynbos* and forest (Table 5.2). In the case of grassland, which is believed to have constituted 24% of the country's land area before 1700 (Rutherford and Westfall 1994; Low & Rebelo 1996), this is due to the disproportionate presence of high impact land uses: In 1995, 47% of South Africa's cultivated land, 47% of urban land, and 56% of plantation areas lay in the grassland biome. In the case of *fynbos* and forest, estimated to have occupied 6% and 0.6% of the country respectively

Table 5.2 A cross-tabulation of BII (%) per major ecosystem type and broad taxonomic group for South Africa. The overall score (81.2 ± 6.9% at the 95% level) can be disaggregated into column or row scores for ecosystems or taxa, or down to taxa within ecosystems. Note that the aggregations are not simple averages, but are weighted by richness and area. The 95% confidence intervals indicate the uncertainty associated with the population impact estimates I_{ijk}.

	Area (km²)	**Plants**	**Mammals**	**Birds**	**Reptiles**	**Amphibia**	**ALL TAXA**	
Richness		*23420*	*258*	*694*	*363*	*111*	***24846***	
Forest	*7 148*	75.6	76.5	88.8	83.7	81.5	**76.1**	(±6.1)
Fynbos	*77 111*	73.1	81.3	93.4	79.0	81.6	**73.7**	(±4.8)
Grassland	*294 815*	71.6	55.5	90.5	75.9	81.0	**73.1**	(±6.4)
Savanna	*416 315*	82.5	71.0	91.5	86.7	93.4	**83.1**	(±8.1)
Thicket	*41 349*	82.7	69.5	91.9	86.9	93.9	**83.4**	(±8.9)
Nama Karoo	*297 806*	87.7	71.3	107.7	94.0	131.1	**89.7**	(±6.0)
Succulent Karoo	*8 2489*	87.4	73.6	106.5	93.1	126.5	**88.3**	(±4.6)
ALL BIOMES	***1 217 033***	**80.2** (±6.6)	**68.6** (±10.1)	**94.9** (±9.4)	**85.7** (±5.7)	**93.6** (±13.2)	**81.2**	(±6.9)

in pre-colonial times (Rutherford and Westfall 1994; Low & Rebelo 1996), the impacts of land use change are amplified by the large number of species affected: 38% of South Africa's species are represented in the *fynbos* biome and 17% of species are represented in the forest biome (Le Roux 2002).

The biodiversity of the Nama and succulent karoo is least impacted, according to our index. We attribute this to their aridity, which limits cultivation and urban settlement. Note that the BII does not include impacts due to climate change, which are projected to be severe in the succulent karoo during the 21st century (Rutherford *et al.* 2000). The savanna and thicket biomes, which together made up about 38% of the pre-colonial land area, have experienced an intermediate level of impact. They are particularly affected by degradation: In 1995, 68% of degraded land in South Africa lay in these biomes. Degradation is defined here as uses that modify, but do not replace, the natural land cover, resulting in a reduction of the productive potential of the ecosystem that persists from year-to-year. In the high-resolution satellite images used for national land cover mapping, the degraded areas show up as brighter than their undegraded neighbours. The most widespread causes of degradation are thought to be long-term grazing at levels far in excess of the productive potential of the ecosystem, and the harvest or browsing of trees at rates greater than they can regrow. Both are often associated with soil erosion (Hoffman & Ashwell 2001).

Mammals are the taxonomic group most affected by human activities in South Africa (Table 5.2). Disaggregated to the level of functional groups (i.e. guilds), it is the very large herbivores and predators that are particularly affected. Species in these two groups pose a direct threat to humans, as well as competing with domestic livestock, and are therefore excluded from almost all areas not managed specifically for wildlife conservation. The populations of the majority of other mammals species, and most birds, plants, reptiles and frogs, are estimated to be only marginally reduced, if at all, in the extensive areas of South Africa used for grazing by domestic livestock. Certain groups specifically benefit from human activities in livestock ranching areas, particularly from the presence of farm dams and artificial water points. This beneficial effect is especially pronounced for amphibian families such as the *Bufonidae*, and explains the high scores for amphibians in the karoo areas (Table 5.2).

Gauteng is the province with the lowest BII score, due to its high level of urban development (Figure 5.1a). The Free State is particularly impacted by cultivation, while plantations are a major impact in Mpumalanga. Degradation is an important factor in the North West and Limpopo

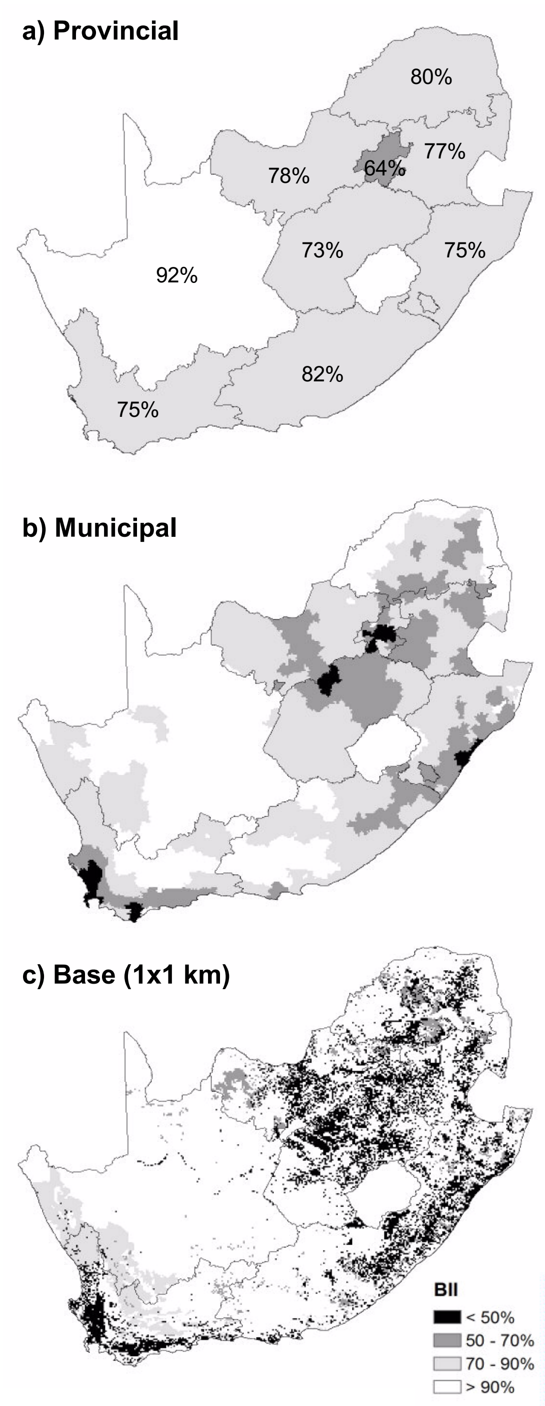

Figure 5.1 The Biodiversity Intactness Index applied at sub-national levels of environmental decision-making in South Africa. Results are richness- and area-weighted averages of BII as estimated at a base resolution of 1x1 km. Values of BII obtained at different scales are directly comparable: they refer to the average abundance of all species in the particular area, expressed as a fraction of pre-colonial era abundance.

provinces, as well as in the Eastern Cape and KwaZulu-Natal. Disaggregation of the BII to local government level allows for within-province comparison and finer-level planning (Figure 5.1b). The high impact of the major metropolitan areas (greater Johannesburg, Cape Town and Durban) is matched by the impacts of cultivation in the region of Bothaville in the northern Free State, and the Bredasdorp region on the Agulhas plain in the Western Cape. Municipalities along the entire seaboard of South Africa, save for a small area near the mouth of the Gariep (Orange) River, have been substantially impacted by human activities, particularly in the Western Cape and KwaZulu-Natal. When BII is calculated at the 1x1 km base resolution of the input data, localised impacts, such as irrigated croplands along the Gariep River in the Northern Cape, become apparent (Figure 5.1c). Note that although we estimate urban development to reduce wild populations within the urbanised area by nearly 90%, the actual footprint of dense human settlement in South Africa is small (1.14 %). Even with high rates of urbanisation, the total area of urban land will remain small for several decades.

This study quantifies the role played by the extensive areas *not* under formal conservation management in maintaining South Africa's biodiversity. The extensive areas used primarily for sheep and cattle ranching, and stocked within grazing norms, contain 80% of the remaining wild vertebrate and plant populations in the country (Table 5.3). This suggests that the national biodiversity framework needs to address the management of these areas, especially in light of the Constitutional mandate to secure ecologically sustainable use of natural resources. While cultivation accounts for over 50% of the present impact on biodiversity, there is limited potential for further expansion of this land use in South Africa. It is estimated that only 13.7% of South Africa is very suitable for crop agriculture, almost all of which is already cultivated (NDA 2004). In contrast, a large fraction of the extensive area (74% of South Africa) currently under moderate extractive uses, primarily livestock grazing, could potentially become degraded if not managed appropriately. On average, we estimate that degradation reduces the abundance of wild populations by 43%; this compares to an average reduction of only 7% in areas under moderate extractive use. The rangelands of South Africa fall under the authority of multiple government departments, including the Department of Agriculture, the Department of Water Affairs and Forestry, and the Department of Environmental Affairs (the lead department for the implementation of the Biodiversity Act). The regulation of the key activities affecting biodiversity therefore calls for effective coordination between government departments.

Table 5.3 Area in South Africa in each land use category, and the mean richness-weighted BII (%) for each category. The 95% confidence intervals (CI) indicate the uncertainty associated with the population impact estimates. The right-hand two columns respectively indicate the percentage of the decrease in populations explained by each land use type, and the percentage of the remaining biodiversity that is now located in each land use type. Cultivation accounts for over half of the current impact on biodiversity. Four-fifths of the remaining wild organisms in South Africa are estimated to be in the extensive areas under moderate extractive use, mostly rangelands. This highlights the importance of proper management of these areas in the conservation of the biodiversity in South Africa.

Land use	Area (km^2)	%	Mean BII (95% CI)	% Decrease in BII	% Remaining BII
Protected	84 750	6.96	100.0 (±0.1)	0.02	11.19
Moderate Use	903 322	74.22	93.3 (±7.9)	24.74	80.40
Degraded	56 909	4.68	56.7 (±11.4)	12.29	3.73
Cultivated	142 766	11.73	25.1 (±4.8)	52.20	4.05
Plantation	15 351	1.26	27.2 (±4.1)	5.70	0.19
Urban	13 935	1.14	12.7 (±4.0)	5.05	0.44
SOUTH AFRICA	**1 217 033**	**100.00**	**81.2 (±0.069)**		

The topic most in need of further research as indicated by the sources of uncertainty in our estimate of BII, is the impact of various land use activities on the populations of native organisms. Our uncertainty is greatest with respect to the impacts of degradation and moderate use (Table 5.3), which are particularly important land use categories in the savanna and thicket. At the taxonomic group level, we are most uncertain about the impact of different land uses on amphibian populations, followed by mammals and birds. More accurate data on the extent and nature of different types of land use will also help to better refine our estimates of BII. The categories which show the biggest variation in South Africa between different land use maps are urban land and degraded land (Biggs *et al.* 2004b). This is largely due to inconsistency in the definition of these categories in different land cover products, together with problems in the detection of specific categories, particularly degradation.

Most freshwater biodiversity, and all marine biodiversity, have been excluded in this study. The BII could, however, be extended to cover these systems. An estimate for marine ecosystems could be done by direct application of the area-based BII algorithm, while linear freshwater and coastal systems may potentially require the substitution of length for area. Species richness data for aquatic plants and vertebrates are available. The main task would be to define impact categories analogous to those presented in Table 5.1, and to collect expert judgements (and/or published field data) regarding the impacts of these categories on the populations of various functional groups. In our experience, this requires several weeks of effort (Scholes and Biggs 2005).

5.4 Conclusion

This study demonstrates the application of a biodiversity assessment tool that can complement the information provided by existing approaches in the development of the South Africa's national biodiversity framework, and in periodic reporting of the state of the country's biodiversity. In particular, the BII highlights the importance of the extensive areas under moderate extractive use in South Africa in maintaining biodiversity in the country. It suggests that the policy action with the greatest potential to limit further loss of biodiversity is to prevent these areas from becoming degraded. Addressing this requires coordination, at all levels, between several arms of government, particularly between the Department of Environmental Affairs and Tourism, the Department of Agriculture and the Department of Water Affairs and Forestry.

The BII complements existing assessment approaches by differentiating between the impacts of different land uses and quantifying the effect of these land uses on the populations of native terrestrial plants and vertebrates. The information provided by the BII about the most impacted biomes and taxonomic groups in South Africa lends independent confirmation to results derived by other means. The advantage of the BII in comparison to existing methods, particularly listings of different land-cover types and protected areas per biome, is that the BII integrates across land uses, biomes, and a broad range of biodiversity to provide a synthetic and sensitive over view of the status of biodiversity.

The findings in this study do not deny the importance of protected areas, particularly to the conservation of threatened species. Indeed, this study would have been impossible if a system of protected areas were not in place to provide a reference baseline. However, in a country where the livelihoods of nearly half of the population depend on the economic use of the landscape, principally through farming crops and livestock, it is unreasonable to expect that a significantly larger fraction than the current protected area can be set aside exclusively for nature conservation. Alongside a policy of selective protection of key areas, there should be a policy to maintain and encourage biodiversity-friendly land use in the remainder of the landscape, where people live.

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Chapter 6: Scenarios of biodiversity loss in southern Africa in the 21st century

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In preparation for submission to *Ambio*

This chapter makes use of the parameterisation of the IMAGE model for the four MA global scenarios in order to model possible changes in land cover in southern Africa over the 21st century, and assess the consequences for biodiversity using the BII. The IMAGE model is one of the leading integrated global environmental assessment models, and includes factors such as energy consumption, population growth, agricultural production and climate change. It is one of the few such integrated models that represents land use on a spatially explicit basis. IMAGE is one of a handful of models that were specifically parameterised for the four MA global scenarios, in order to explore the consequences of plausible trajectories of global development for ecosystems and ecosystem services. A specific feature of the MA scenarios was that they specifically tried to explore the consequences of differences in approaches to environmental management, specifically the differences between proactive and reactive management, for ecosystem services.

This paper represents a collaborative effort between the IMAGE modelling team and the authors who developed the BII. H. Simons was responsible for coordinating the collaborative effort, hence his lead authorship. Certain modifications to the IMAGE land cover model and output were needed in order to apply the BII. Given the complexity of making such modifications in a large model such as IMAGE, several technical members of the IMAGE team were involved in performing the analyses: M. Bakkenes, B. Eickhout, and D. van Vuuren. I played a lead role in defining which modifications were required, and suggesting the ways in which these could be implemented, given the functional relationships coded into IMAGE. R.J. Scholes provided additional input in this respect, and commented on the draft manuscript. I played a lead role in defining the content of the paper and writing and editing the manuscript. My contribution to the paper is recognized by the order in which the authors are listed.

Chapter 6: Scenarios of biodiversity loss in Southern Africa in the 21st Century

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Biodiversity is currently being lost around the world at rates far in excess of the average rate of speciation in the fossil record. To date the biodiversity of southern Africa is relatively intact, but will this situation persist into the 21st century? Agricultural expansion, which is projected to continue well into the coming century, is the primary reason for the current loss of biodiversity in the region. In this paper, we assess the impacts of land use changes, as projected by a leading integrated global environmental assessment model (IMAGE) under a range of scenarios, on the terrestrial biodiversity intactness of southern Africa. The Biodiversity Intactness Index gives the average change in population size, relative to the pre-modern state (pre-1700), across all terrestrial species of plants and vertebrates. We project absolute declines of between 21% and 33% in the average population sizes of these species by 2100. The rate of decline is projected to accelerate between 2020 and 2060 under all scenarios. However, under both scenarios of high and low economic growth, pro-active approaches to ecosystem management significantly reduce biodiversity impacts.

6.1 Introduction

The current loss of biodiversity, at rates significantly exceeding those in the fossil record (Lawton & May 1995; Anderson 2001), is a source of concern to many people, conservation organisations and countries. Biodiversity loss can entail any or all of the following: reduction in the extent or productivity of ecosystems; declines in the abundance or distribution of populations of particular species; and loss of genetic diversity within populations (CBD 2003). Biodiversity plays a key role in the functioning of ecosystems and the provision of ecosystem services, and its loss therefore impacts on human well-being (MA 2005).

Much research has focused on the factors underlying the accelerated loss of biodiversity in the 20th century. The major anthropogenic drivers are land use change, nitrogen deposition and climate change (MA 2005; MA *in press*). Several studies have investigated the impacts of climate change. They conclude that climate change is already affecting species distributions all over the world (Parmesan and Yohe 2003), and will have considerable further impacts in the 21st century (Gitay *et al.* 2002; Thomas *et al.* 2004; Leemans and Eickhout 2004). All models project a several degree temperature rise in southern Africa, and many models project a decrease in precipitation during summer (IPCC 2001).

For terrestrial ecosystems, land use change is expected to be the dominant driver of biodiversity loss over the next century (MA 2005; Sala *et al.* 2000). Land use change is mainly driven by agricultural expansion. Agricultural expansion is, in turn, driven by population growth and the shift to higher protein diets which accompanies increased household income (Sage 1994). Consequently, a growing fraction of the expanding agricultural area is used for the production of food and fodder crops for feeding animals (Bouwman *et al.* 2004). The bulk of agricultural expansion in the 21st century is expected to occur in sub-Saharan Africa, and particularly in Angola, the Democratic Republic of Congo, and Sudan (Alcamo *et al.* 1998; Bruinsma 2003). These are among the major remaining areas of the Earth's surface that have high agricultural potential, but have not yet been extensively cultivated. Since southern Africa is one of the highly biodiverse regions of the world (WCMC 2000), the impact of climate and land use change on global biodiversity has the potential to be disproportionately high.

The objective of this paper is to explore the impacts of several scenarios of land use and climate change on the biodiversity of southern Africa over the 21st century. For the purposes of this paper, southern Africa is defined as the region comprising Angola, Botswana, Lesotho, Malawi, Mozambique, Namibia, South Africa, Swaziland, Tanzania, Zambia and Zimbabwe, with a total area of about 7 million km^2. We use the Integrated Model to Assess the Global Environment (IMAGE) (Alcamo *et al.* 1998; IMAGE-Team 2001) to model future scenarios of land use and climate chance in the region. The consequences of these scenarios for terrestrial biodiversity are assessed using the Biodiversity Intactness Index, which has been proposed for use in monitoring progress towards the Convention on Biological Diversity's (CBD) target of reducing the rate of biodiversity loss by 2010 (Scholes and Biggs 2005).

6.2 Scenarios of land use change

Future trends in climate and land use are the product of complex dynamic processes driven by demographic change, socio-economic development, and technological and institutional change. The development of storyline-based scenarios, which are subsequently quantified using large integrated models, has become a popular tool to explore possible trajectories of change (Nakicenovic & Swart 2000; UNEP 2002; MA *in press*). The most recent of these initiatives, the Millennium Ecosystem Assessment (MA), focused specifically on quantifying changes in ecosystem services, and was quantified by a selection of large integrated models. The MA scenarios provide the greatest detail and consistency in terms of integrated drivers of future land-use change in the region, and were therefore used in this study.

Two major axes define the MA scenarios (MA *in press*): i) a continuum from strong globalisation to strong regional development; and ii) a continuum from a reactive to a proactive attitude towards ecological change and ecosystem management. Four corresponding scenarios were developed (Table 6.1). The *Global Orchestration* scenario describes a world with a strong focus on social and economic development, but with less emphasis on ecological protection. Economic growth and technological development are relatively fast, and the global population peaks at slightly more than 8 billion people in 2060 as a consequence of a fast decline in fertility levels. The *Techno Garden* scenario also describes a globalized world, with relatively rapid economic growth and a relatively low population trajectory. However, here a strong emphasis is placed on protecting the ecological services provided by ecosystems. In the *Order from Strength* scenario, regions

Table 6.1 Key features of the four Millennium Assessment scenarios (MA *in press*).

GLOBALIZED

Global Orchestration

Potential Benefits

- Economic prosperity and increased equality due to more efficient global markets
- Wealth increases demand for a better environment and the capacity to create a better environment

Inadvertent Consequences

- Progress on global environmental problems may be insufficient to sustain local and regional ecosystem services
- Breakdowns of ecosystem services create inequality (disproportionate impacts on the poor)
- Reactive management may be more costly than preventive or proactive management

Techno Garden

Potential Benefits

- Highly efficient management and utilization of ecosystems
- Technological enhancement of ecosystem services
- Forward-looking market mechanisms efficiently allocate ecosystem services

Inadvertent Consequences

- Increasing reliance on particular technologies may decrease the diversity of systems for providing ecosystem services, thereby increasing vulnerability to surprising breakdowns
- Wilderness disappears as "gardening" of nature increases, and people have fewer experiences of nature
- Less economic growth because of diversion of resources to environmental technology

REACTIVE

PROACTIVE

Order from Strength

Potential Benefits

- Increased security for those who can afford it
- Political and trade barriers slow the spread of invasive species and some diseases
- Some regions remain well-endowed with ecosystem services

Inadvertent Consequences

- Lower economic growth because of fragmentation, inequality, conflict, and lost human potential
- Environmental degradation of the global commons, and losses of ecosystem services in poor regions
- Vulnerability due to fragmentation of ecosystem services

Adapting Mosaic

Potential Benefits

- Integration of management institutions with ecological processes to improve the resilience of ecosystem services
- Growth of adaptive capacity to sustain ecosystem services in a changing world

Inadvertent Consequences

- Little progress on global ecosystem problems
- Less economic growth than max possible because of regionalization of economies and inefficiencies of experimentation

REGIONALIZED

emphasise their cultural identity and most of the development takes places in separate regional blocks. Technological and economic development in this scenario is slower than in the globalized scenarios, while security forms a dominant theme in policy-making. Finally, in *Adapting Mosaic*, regional communities decide to rediscover their relationships with their local environment. Development is therefore focused at the regional level, with a medium economic growth path.

These four scenarios result in significant differences in human populations and wealth (expressed as gross domestic product (GDP) per capita) in southern Africa by 2100 (Figure 6.1). In the two globalisation-oriented scenarios, *Global Orchestration* and *Techno Garden*, the regional GDP rises at about 3% per year, whereas in the more regionalized *Order from Strength* scenario the growth is 1.2% per year. GDP growth in the self-reliant *Adapting Mosaic* scenario lies between these extremes. The regional population peaks during the 21st century in all scenarios, at twice the current population in *Global Orchestration* in 2070, and at three times the current population in *Order from Strength* in 2090. As a result, individual wealth rises tenfold in the former scenario, whereas it hardly increases over current levels in the latter. These changes in total population and individual wealth are the key drivers of the pattern of land use change that evolves under

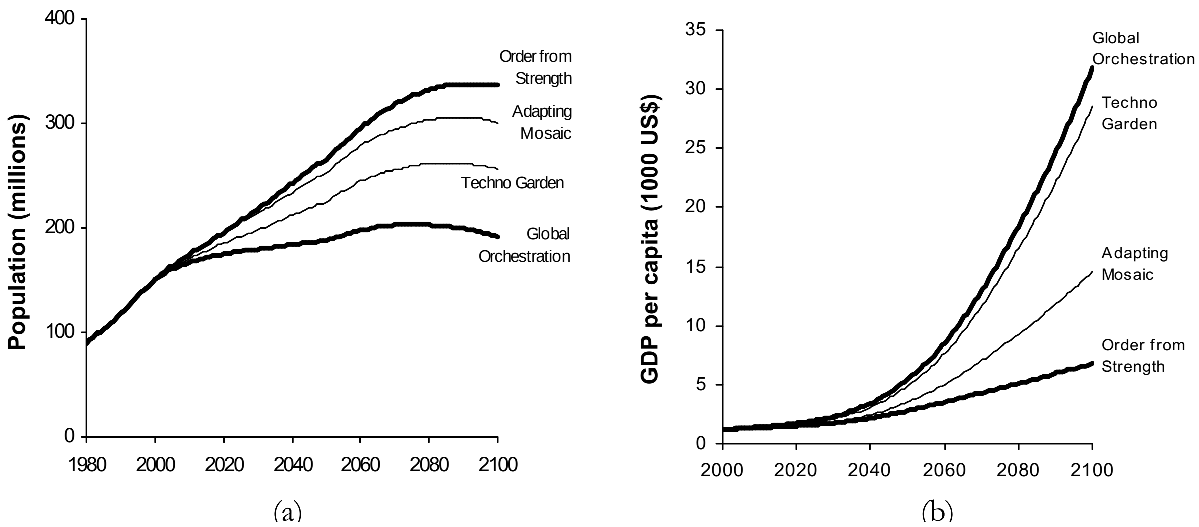

Figure 6.1 Trends in population (a) and GDP per capita (b) for the period 1980-2100 under the four MA scenarios. Scenarios with strong regionalized global development (*Order from Strength* and *Adapting Mosaic*) have higher total populations and substantially lower economic growth.

the various scenarios. The higher per capita human wealth, the more protein-rich diets tend to be, leading to a higher demand for animal products. Technological transfers reach more wealthy regions first leading to intensified agricultural practices ('intensification'), while less wealthy societies increase food production predominantly by enlarging the cultivated area ('extensification').

6.3 Data and methods

The four MA scenarios were quantified by a selection of large integrated models. The IMAGE model provided the most detail on future land use and land cover change under the MA scenarios, and was therefore selected for this study. While several other detailed studies of future land use change in sub-Saharan Africa exist, they have tended to have a narrower focus, for instance focusing on climate change impacts on agriculture (Fischer *et al.* 2002), rather than the broader spectrum of drivers considered in the MA scenarios and of interest in this study.

The* IMAGE 2.2 *Modelling framework

IMAGE 2.2 consists of a set of linked and integrated models, which together describe important elements of the causal chain of global environmental change. The framework and its sub-models have been described in detail (Alcamo *et al.* 1998; IMAGE-Team 2001; Leemans and Eickhout 2004). IMAGE has been used for several important global scenario analyses: IPCC's Special Report on Emission Scenarios (Nakicenovic & Swart 2000); UNEP's Global Environment Outlook (UNEP 2002) and the Millennium Ecosystem Assessment (MA *in press*). IMAGE is one of the few large integrated global models that represents land use on a spatially explicit basis and integrates environmental drivers such as climate change. Other important elements of IMAGE include emissions of greenhouse gasses and regional air pollutants. In the model, socio-economic processes are mostly modelled at the level of 17 world regions, while climate, land-use and several environmental parameters are modelled at a 0.5 by 0.5 degree resolution.

IMAGE models the global-mean temperature increase based on emissions of greenhouse gases. Climate-change patterns are not simulated explicitly in the IMAGE-model.

Instead, the global mean temperature increase is linked to climate patterns generated by a general circulation model (GCM) for the atmosphere and oceans. This linking takes place using the standardised IPCC pattern-scaling approach (Carter *et al.* 1994) and additional pattern-scaling for the climate response to sulphate aerosols forcing (Schlesinger *et al.* 2000). The implementation of these approaches in the IMAGE-model are described in detail by (Eickhout *et al.* 2001). GCMs are currently the best tools available for simulating the physical processes that determine global climate dynamics and regional climate patterns.

The land-use sub-model of IMAGE distinguishes 14 natural land-cover types (biomes) and 4 anthropogenic land-cover types. The approach used for modelling the potential distribution of crops and crop yields is similar to the FAO's agro-ecological zones approach (Leemans and Van den Born 1994). The BIOME model (Prentice *et al.* 1992) is used to determine the potential distribution of biomes, and biome patterns are updated every five years. Vegetation patterns change as a result of climate change, the atmospheric CO_2 concentration, and migration of species. BIOME calculates an instantaneous equilibrium response to climate change by shifting biome patterns. In reality such shifts take time. Grasses migrate more rapidly than long-lived trees. The migration process is a function of the rate of climate change, original and new vegetation types, and the distance to the nearest location where the new vegetation type already exists (Van Minnen *et al.* 2000).

Changes in the anthropogenic land-use classes are driven by demands for food, (including crops, feed, and grass for livestock), timber and energy crops. For the expansion of agricultural land, a rule-based 'suitability map' determines which grid cells are selected. Conditions that enhance the suitability of a grid cell for agricultural expansion are its potential crop yield, its proximity to other agricultural areas and its proximity to water. Like the potential vegetation, the potential crop yield changes over time because of climate change.

The Biodiversity Intactness Index

The BII is an aggregate index which is intended to provide a high-level overview of the state of biodiversity for the public and decision-makers (Scholes and Biggs 2005). It was developed for use in the Southern African Millennium Ecosystem Assessment (Scholes & Biggs 2004; Biggs *et al.* 2004) to provide a synthetic assessment of biodiversity status in the region at several spatial scales. The BII combines information on ecosystem distribution, species richness and the extent and impact of major land uses. The BII can be calculated using existing data, while being amenable to the use of more detailed or higher quality data sources in future. An error bar can be associated with the BII, and the long-term research and observation goal can be defined in terms of shrinking the uncertainty range. In addition to the high-level synthetic score, the BII can be disaggregated spatially (by ecosystem or political boundaries), by taxonomic group or by human activity to meet the more detailed information needs of other users.

The full mathematical derivation of the BII is given in Scholes and Biggs (2005). The BII is formally defined as the average, across all species of well-known terrestrial biodiversity (plants, mammals, birds, reptiles and amphibians), of their populations at a particular time relative to their populations at a reference time. The reference in this study was nominally the pre-colonial period (i.e. pre-1700); in practice contemporary abundances in large protected areas were used as a proxy. The BII is calculated as the average population impact (I_{ijk}) of the various human activities that make up the terrestrial landscape, weighted by the extent of each activity (A_{jk}) and the species richness of the areas affected (R_{ij}):

$$BII = \frac{\sum_i \sum_j \sum_k \mathrm{R_{ij} A_{jk} I_{ijk}}}{\sum_i \sum_j \sum_k \mathrm{R_{ij} A_{jk}}}$$

where

I_{ijk} is the population of taxon i under land use activity k in ecosystem j, relative to a reference population in the same ecosystem type;

R_{ij} is the richness (number of species) of taxon i in ecosystem j; and

A_{jk} is the area of land use k in ecosystem j.

Species-by-species data for estimating I_{ijk}, the impact of a particular land use activity on a particular group of organisms, is seldom available, particularly in many of the highly biodiverse, developing regions of the world. Scholes and Biggs (2005) therefore used a panel of sixteen experts to independently estimate the change in population abundances of five taxa (plants, mammals, birds, reptiles, and amphibians) for a range of human land use activities (Table 6.2). Estimates were made for each of about ten functional types per broad taxon, for each of six major biome types (forest, savanna, grassland, arid shrublands, *fynbos* and wetland) in southern Africa. These estimates were aggregated up to the broad taxon level by weighting by the species richness of each functional type.

Species richness (R_{ij}) is typically available as the richness per broad taxonomic group, per ecosystem type. In this study, species richness data (WWF-US 2003) associated with the WWF Ecoregions (Olson *et al.* 2001) were used. A_{jk}, the area of a particular land use within a specific ecosystem type, is determined by overlaying land use and ecosystem maps. In this study, the IMAGE model was used to calculate A_{jk} by relating the IMAGE land cover classes to the land uses and biomes defined by Scholes and Biggs (2005) for collecting the I_{ijk} population impact values (Table 6.2).

Modelling scenarios of biodiversity intactness using IMAGE

Changes in the BII of southern Africa over the 21st century were assumed to be a function of changes in the area under different forms of land use and shifts in the distribution of biomes, i.e. changes in A_{jk}. Changes in land uses and biomes were modelled at a decadal time-step under the four MA scenarios using the IMAGE model. Changes in land use projected by IMAGE result from the combined effects of many variables, including population growth, climate change and changes in the intensity of agricultural practices. The impact of specific types of land use on the abundance of wild organisms (I_{ijk}) was assumed to remain unchanged during the simulation period, and no new land uses were introduced. Since species richness data were not available at the biome level, species richness per biome was taken as the average richness of each taxon per ecoregion (Olson *et al.* 2001; WWF-US 2003).

Table 6.2 Land use classes used to model changes in biodiversity intactness (BII). Existing IMAGE land cover classes were related as closely as possible to these land use classes in order to calculate the BII.

Land use	Description	Corresponding IMAGE classes	BII (averaged across taxa and biomes)
Urban	Dense urban and industrial areas, mines and quarries	No corresponding IMAGE class	14.2%
Plantation	Plantation forestry, mainly pine and eucalypt species	No corresponding IMAGE class	26.8%
Cultivated	Commercial and subsistence crop agriculture, and intensive 'industrial' grazing	Agricultural land (cropland fraction and a portion of the grassland fraction). Also fraction of the extensive grassland class in cases where the agricultural land class cannot accommodate the full extent of the projected cultivated area.	28.1%
Degraded	Clear-cut logging, areas subject to intense harvesting, hunting or fishing, areas invaded by alien plants	Regrowth areas (after abandonment or timber cut) classes. Fractions of the agricultural land, extensive grassland, warm mixed forest, grassland and steppe, hot desert, scrubland and savanna classes	52.8%
Moderate use	Forest areas used by indigenous people, grasslands grazed within their sustainable carrying capacity	All remaining areas not classified into one of the other five BII land use categories	92.7%
Protected	Large protected areas	Bioreserves	100%

For the purposes of this study, a new IMAGE biome, *fynbos*, was defined as a subset of the IMAGE scrubland class to match the biomes defined by Scholes and Biggs (2005). The extent of *fynbos* was determined by the overlap between the IMAGE scrubland class and the *fynbos* class mapped by Low and Rebelo (1996). This means that the maximum extent of *fynbos* as applied in this study is equal to or smaller than the area mapped by Low and Rebelo (1996). This is justified by the fact that *fynbos* vegetation is restricted to the nutrient poor mountainous areas of the South-western Cape Province of South Africa, and is not projected to undergo substantial changes in areal cover as a result of climate change (Rutherford *et al.* 2000).

Several land use classes defined by Scholes and Biggs (2005) for calculation of the BII are not represented, or only partially represented, in the IMAGE model. These include the protected, urban and degraded classes. IMAGE represents only international bioreserves and not other state- and privately-owned reserves, and the area of bioreserves does not change over time or differ between scenarios. Given the steep rise in protected area in southern Africa during the 1990s, the lack of a dynamically-changing protected area class is a limitation of the current version of the IMAGE land cover model. The expansion of protected area in the region is, however, expected to saturate in the early 21st century at between 10% and 15% of the total land area, so the potential effect on our results is constrained. Similarly, IMAGE does not incorporate an urban land use class, but again the total area in question is relatively small (less than 5%) and the potential effect on our results thus limited.

The degraded land use class is also not directly represented in IMAGE, but because degradation is expected to potentially have major impacts on biodiversity in the region (Scholes and Biggs 2005), an attempt was made to incorporate it as a dynamic land use class in this analysis. In addition to the two IMAGE classes that were defined as degraded (Table 6.2), the degraded fraction within the other classes was determined by overlaying the BII land use map derived by (Scholes and Biggs 2005) on the IMAGE land cover map. The degraded fraction in each of these classes was adjusted over time in proportion to the population trend in the region, which differed by scenario.

The extent of the IMAGE agricultural land class in the base simulation is substantially larger than the current extent of cultivation as mapped on satellite-derived land cover maps. The IMAGE agricultural land class was therefore subdivided into a cropland class and a grassland class using the fraction of grassland within each cell, as given in the IMAGE output. The cropland area was assigned as cultivated. The grassland area could be either intensively used, for example to grow fodder for livestock, or extensively used as rangeland for livestock grazing. Based on Bouwman *et al.* (2004), we constructed a relationship between the amount of grass consumption per hectare and the grazing intensity (Figure 6.2). Using this relationship and IMAGE data on grass consumption and grassland area, we calculated and projected the area of intensively and extensively used grassland under each MA scenario. The intensively used grassland was assigned to the cultivated land use class. The extensively used grassland was designated as moderate

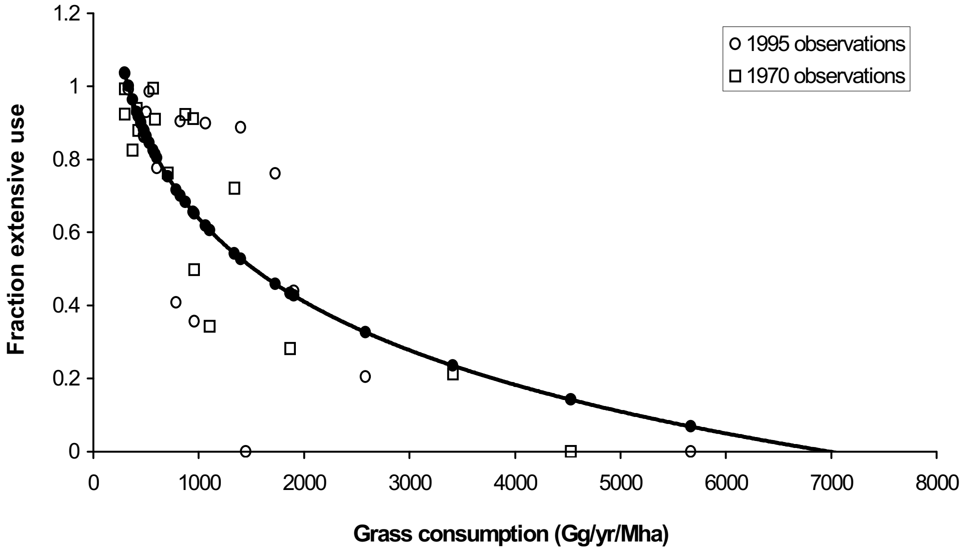

Figure 6.2 The relationship between grass consumption by livestock (grazing intensity) and the fraction of grassland extensively used, based on IMAGE data for 1970 and 1995. The fraction of grassland under extensive use is given by the following equation: Fraction extensively used grassland = 2.9075 – 0.3285 * LN(grass consumption). The fraction of intensively used grassland is simply the inverse fraction.

use and distributed over the remaining area of the IMAGE agricultural land and extensive grassland classes. Where the remaining area in the agricultural land class was too small to accommodate all the required cropland and/or intensively used grassland, the remainder was allocated in the IMAGE extensive grassland class. All the unassigned 'natural' area was classified as moderate use.

6.4 Results and discussion

A substantial and ongoing loss of biodiversity is projected for Southern Africa under all four MA scenarios during the 21st century (Figure 6.3). The BII in 1995 was 89%. BII estimates for 2100 range between 68% in the regionalized, proactive *Adapting Mosaic* scenario and 56% in the globalized, reactive *Global Orchestration* scenario. The absolute decline between 2000 and 2010 is estimated as 1.4% under the *Techno Garden* scenario, and 1.3% under all other scenarios. This is higher than the rate of 0.8% estimated by Scholes and Biggs (2005) for the decade of the 1990s for the southern part of the region, suggesting that the CBD target of reducing the rate of biodiversity loss by 2010 will remain out of reach. Under all scenarios, the decline in biodiversity is projected to accelerate between about 2020 and 2060, with absolute rates ranging from 2.3% to 5.0% per decade. Under the *Adapting Mosaic* scenario the rate of biodiversity loss falls to 0.2% per decade after 2080. Under the other scenarios, absolute rates of declin e range between 1% and 2% per decade by the end of the 21 st century. Under no scenario does a decadal increase in BII occur before 2100.

The major driver of the ongoing decline in biodiversity is agricultural expansion and the intensification of livestock production (Figure 6.4). The projected land use changes are triggered by an increasing demand for crop products, induced by human population growth, and a rising demand for animal products linked to a rise in per capita GDP (Figure 6.1). All scenarios project a rapid increase in the total production and consumption of animal products, most strongly in the wealthy globalized scenarios (*Global Orchestration* and *Techno Garden*). Although the majority of livestock in the region are currently raised on natural vegetation (savanna, grassland and shrublands), which has a relatively minor impact on biodiversity, the capacity for increased production from these 'extensive' practices is limited, particularly in the nutrient-poor woodlands that cover the moister part of the study area. Therefore, satisfying the projected increase in demand will require a substantial increase in livestock raised on planted and fertilised

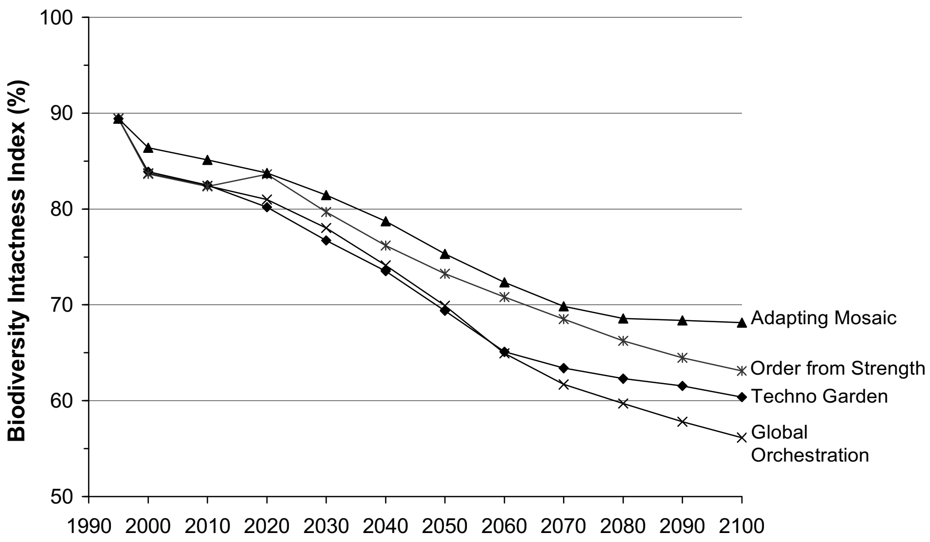

Figure 6.3 BII trends 1995-2100 under the four MA scenarios. The lowest BII in 2100 is projected under the globalized, reactively managed *Global Orchestration* scenario, where per capita incomes are highest and total population lowest (Figure 6.1). The least loss in BII is projected under the regionalized, proactively managed *Adapting Mosaic* scenario, where the total population is relatively large and income per capita relatively low.

pastures cleared from woodlands, or on feeds derived from cultivation, such as maize and soybeans. The relatively higher consumption per capita and greater shift to intensively produced protein-rich foods, underlies the large cultivated area (making up about half the land area by 2100), and consequent high loss of biodiversity projected under the *Global Orchestration* and *Techno Garden* scenarios (Figure 6.3). This is despite the fact that production per unit land is higher than under the *Order from Strength* and *Adapting Mosaic* scenarios. The low-caloric preferences under the *Techno Garden* scenario explain the smaller fraction of cultivated land compared to the *Global Orchestration* scenario.

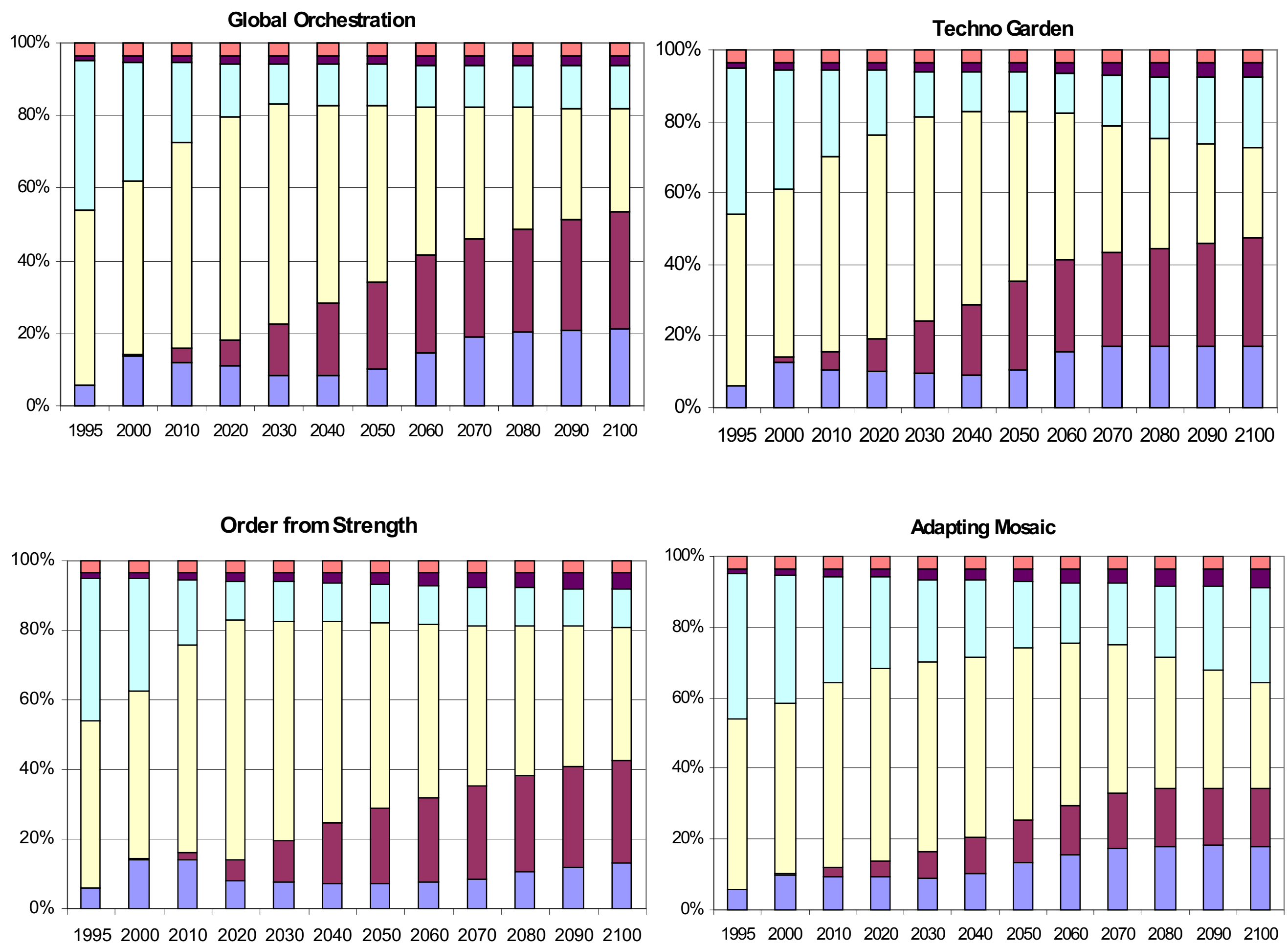

Figure 6.4 Trends in land use 1995-2100 under the four MA scenarios. Cultivated area, in the form of cropland and planted pastures for livestock fodder, is the major driver of the projected biodiversity loss in southern Africa over the 21st century. Protected area remains constant throughout the simulation period under all scenarios, at 3.5% of the total land area of 6 763 854 km². The area of degraded land follows trends in human population (see methods).

In the two scenarios with a strong regionalized global economy and lower economic growth (*Order from Strength* and *Adapting Mosaic*), the total cultivated area is smaller, and the impact on biodiversity by 2100 consequently less. This situation is driven by the larger, but poorer human populations expected under these two scenarios (Figure 6.1), so that fodder production for livestock makes up a smaller fraction of the total cultivated area. Instead, a greater fraction of livestock are raised on natural vegetation by extensive practices. Under *Adapting Mosaic* a larger fraction of the cultivated area consists of crops for direct human consumption compared to the two globalized scenarios (Figure 6.4). Despite the greater per capita wealth in the *Adapting Mosaic* scenario, the area devoted to the cultivation of fodder crops is lower than in the *Order from Strength* scenario. The stabilising area of cultivated land under the *Adapting Mosaic* scenario explains the declining rate of biodiversity loss in that scenario after 2070 (Figure 6.3).

Differences between biomes

Projected differences between biomes by 2100 are larger than the overall differences between scenarios (Table 6.3). There is a clear correlation between areas of high agricultural potential and high biodiversity impact. Biodiversity remains high in the driest biome, shrubland, which is unsuitable for cultivation. All other biomes experience substantial biodiversity losses by 2100, with declines in BII as high as 45% (absolute) in the forest biome under the *Global Orchestration* scenario. A BII score below 40% is projected for the grassland biome by 2100 under the *Techno Garden* scenario on account of the high agricultural potential of this biome. The projected impacts on the *fynbos* biome also deserve particular emphasis as the *fynbos* is limited in extent, covering only 0.6% of Southern Africa, and has remarkably high plant diversity.

Two main effects of climate change on the areal extent of biomes are apparent by 2100: an expansion of the forest biome southwards, and a simultaneous expansion of the arid shrubland biome northwards. Both these expansions occur largely at the cost of the savanna biome. The apparent contradiction of expansion in both the wettest and driest biomes is consistent with the 'consensus' climate model projection of more plant-available water in the northern and eastern parts of the subcontinent in the 21st century, but less in the south-western parts. The shifts in biomes are least pronounced under *Techno Garden*, the scenario with the lowest projections of climate change.

Table 6.3 BII (%) in 2100 compared between biomes and scenarios.

	Baseline 1995		*Global Orchestration*		*Techno Garden*		*Order from Strength*		*Adapting Mosaic*	
	% Area	*BII*	*% Area*	*BII*	*% Area*	*BII*	*% Area*	*BII*	*% Area*	*BII*
Fynbos	0.6	78	0.6	48	0.5	42	0.6	51	0.5	45
Forest	1.1	82	7	37	5.3	41	6.8	48	6.2	46
Savanna	86.3	90	74.2	54	78.5	59	74.9	61	75	68
Shrubland	9.2	93	15.7	92	12.9	92	15.2	91	15.7	92
Grassland	2.7	78	2.5	42	2.8	38	2.5	50	2.5	41
All	**100**	**89**	**100**	**56**	**100**	**60**	**100**	**63**	**100**	**68**

Comparable scenario analyses of biodiversity loss in southern Africa (Sala *et al.* 2000; MA *in press*) have been carried out as part of global scale analyses. Although there are differences in biome classification and regional boundaries between the studies, the broad findings are consistent with our results. Regarding variations across biomes, Sala *et al.* (2000) reported that Tropical forest, savanna and the Mediterranean biome, of which *fynbos* is a subtype, show substantial impacts from land use. The MA global biodiversity scenarios also project large habitat losses, and subsequent loss of biodiversity, in savanna, tropical forest and scrub (including *fynbos*), particularly in Africa (MA *in press*).

Differences between countries

Countries in the eastern and northern parts of the region (Tanzania, Mozambique, Lesotho and Swaziland), undergo substantial transformation by 2100 under all scenarios, while Namibia, Botswana and the western half of South Africa remain relatively untransformed on account of their aridity (Figure 6.5). The extent of transformation in Angola and Zambia differs considerably between scenarios, ranging from modest transformation under the *Adapting Mosaic* scenario to substantial transformation under the *Global Orchestration* scenario. Larger areas with a high BII score remain under the scenarios with proactive ecosystem management (*Adapting Mosaic* and *Techno Garden*). Although the projected BII in 2100 is lower under the *Techno Garden* scenario than under *Order from Strength*, the proportion of relatively untransformed area is higher. The low levels of wealth under *Order from Strength* limit the application of technology and inputs required for high yield agriculture; consequently a larger area is used less intensively.

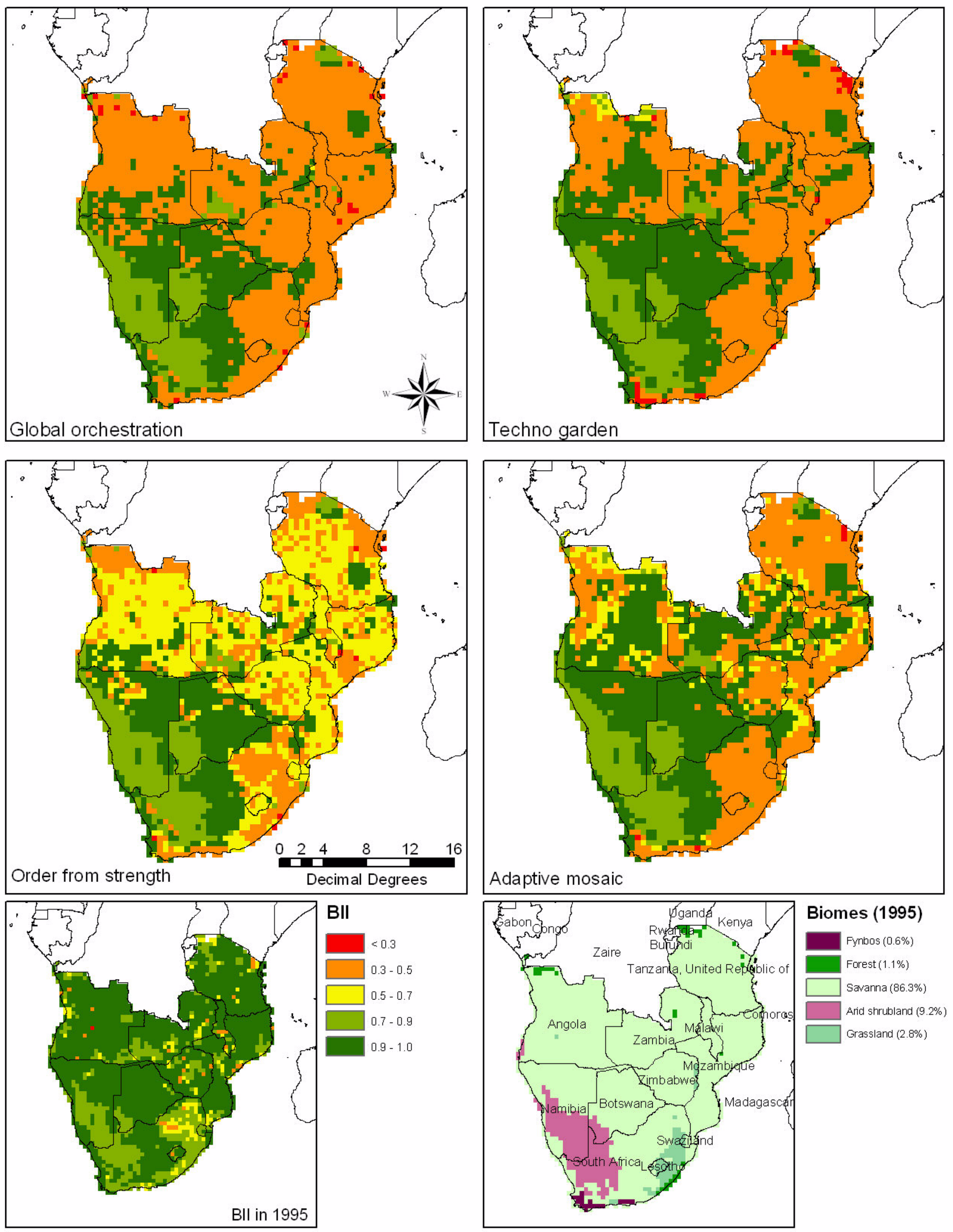

Figure 6.5 BII in 2100 under the four MA scenarios. The total area with low biodiversity impact is largest under the scenarios with a proactive attitude to ecosystem management (*Techno Garden* and *Adapting Mosaic*). The high intensity of agricultural land uses in the wealthy scenarios (*Global Orchestration* and *Order from Strength*) mean that the areas that are transformed have a high biodiversity impact.

Differences between taxonomic groups

The projected changes in land use have differing impacts on the different taxonomic groups. Conversion to agriculture removes most of the original, natural vegetation, although a fraction of the original biodiversity persists in the spaces between fields. Plant population declines are roughly proportional to the planted area. Changes in reptile and amphibian populations tend to follow those of plants due to the relatively restricted size of the habitat required by each individual organism. Land use changes have the largest impact on mammal populations, because of fragmentation of habitat and the incompatibility of large herbivores and carnivores with crop agriculture and livestock husbandry. Birds are least affected by land use changes, and their populations remain high even in the face of extensive transformation (Figure 6.6).

Our results show similar trends to those found in other studies of biodiversity loss. The MA specifically modelled loss of vascular plant biodiversity resulting from habitat loss, using the species area approach (SAR) (MA *in press*). For the Afrotropic realm, roughly covering the region of sub-Saharan Africa, relative projected species loss between 1995 and 2050 ranged from 16% in *Adapting Mosaic* to 33% in *Order from Strength*. This

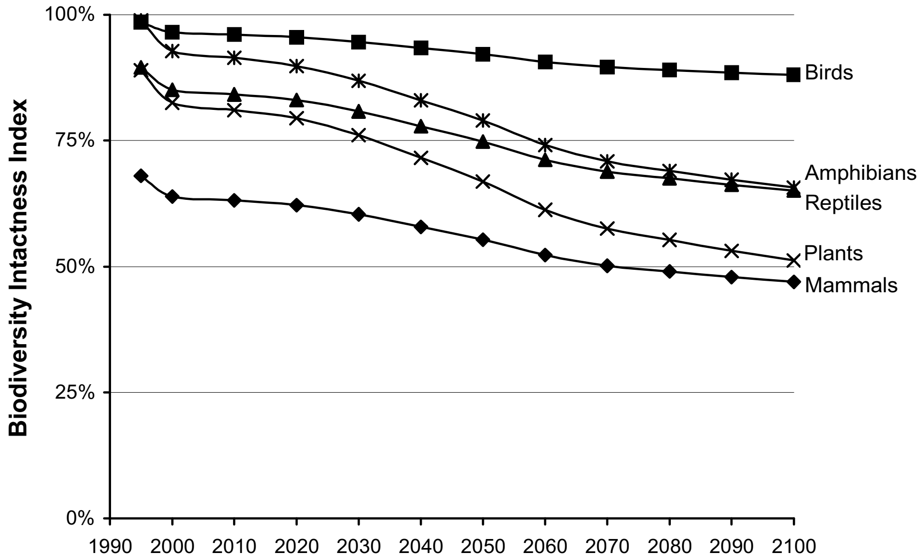

Figure 6.6 Trends in BII 1995-2100 for the broad taxonomic groups, illustrated for the *Global Orchestration* scenario where the total BII loss is greatest. Similar trends are apparent in the other scenarios, although the overall losses are lower.

compares to our range of a 19% decline in the average *abundance* of individual species of vascular plants in the *Adapting Mosaic* scenario to a 27% decline in the *Techno Garden* scenario over the same period.

Limitations of the analyses

Our analysis of biodiversity loss is primarily based on the impacts of projected land use change. Our approach goes a step further than estimating habitat loss as we take into account the differential impacts of the various intensities of land use. Furthermore, we focus on changes in the *abundance* of individual species, rather than projecting the *extinction* of species. Impacts of climate change were indirectly taken into account, through biome shifts and changes in crop productivity as projected in IMAGE. However, the impacts of climate change on species composition and the abundance of individual species within a particular biome was not modelled. The long-term impact of factors such as nitrogen deposition, pollution and invasive species were also not considered, although their immediate impacts, for instance in cultivated areas, were taken into account in the estimates of I_{ijk}. Were these long-term factors taken into consideration, they would lead to higher projections of biodiversity loss than given by considering only immediate impacts of these factors.

Our method could be refined by improving the definition, resolution and causal pathways in modelling changes in the various land use classes. This would entail recoding of parts of the IMAGE model in a subsequent version. In particular, protected and urban areas could be dynamically included in the model. Furthermore, the definition and causal pathways of intensively and extensively used grazing land could be improved, especially for the southern African context where a major fraction of the grazing land consists of natural vegetation rather than planted pastures. Similarly, improving the definition and causal pathways for modelling degraded area would improve the usability of the IMAGE land cover outputs for purposes of assessing biodiversity impacts.

6.5 Conclusions

Our results confirm that land use changes, primarily as a result of agricultural expansion, is likely to be a major driver of biodiversity loss in southern Africa during the 21st century. Projected losses are considerable under all four scenarios, but most marked in the two scenarios where the world becomes increasingly globalized and southern Africa experiences strong economic growth. This occurs despite the lower population levels under these two scenarios, and results primarily from increased food consumption per capita, particularly of animal protein, which requires relatively large land areas. The converse, however, does not hold: the scenario with the lowest levels of per capita wealth does not experience the least impact on biodiversity. Under conditions of low economic growth and low modernisation, the human population in the region continues to grow rapidly, and given the lack of capital for agricultural intensification, extensive areas of southern Africa are transformed to low-yield agricultural uses. Agricultural expansion and the consequent loss of biodiversity thus emerge from an interaction between total population size and per capita economic wealth.

In a globalized and a regionalized world, scenarios in which a proactive attitude to ecosystem management is adopted, lead to lower losses in biodiversity than scenarios in which ecosystem management is reactive. Our findings stress the need for conservation measures, particularly for the *fynbos*, forest and grassland areas. These biomes are characterised by unique and endemic biodiversity, and their extent in southern Africa is limited, particularly in the case of *fynbos*. All scenarios foresee significant land use changes and subsequent biodiversity losses in these areas over the 21st century.

The CBD target of significantly reducing the rate of biodiversity loss by 2010 appears infeasible for southern Africa under the range of scenarios explored in this study, and particularly in light of the Millennium Development Goals for reducing poverty in the region. Our results indicate an increasing rate of biodiversity loss between 2020 and 2060 for all scenarios considered in this study. This does not imply that targets that aim to limit the loss of biodiversity are useless. Rather, it implies that such targets need to take account of the current level of economic development and human development targets in the region. Some loss of biodiversity appears unavoidable if human well-being in southern Africa is to be improved, but environmental management can have a significant influence on the efficiency with which development occurs at the expense of the environment.

6.6 Acknowledgements

The experts who contributed to the BII impact score estimation are thanked for their time and willingness to share their expertise. We thank Neil Burgess, Jennifer D'Amico and the Conservation Science Programme at WWF, and Gerold Kier and the Biomap Working Group at the University of Bonn, Germany for species richness data.

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Chapter 7: Historical changes in natural capital in South Africa: an approach using changes in biodiversity

R. Biggs and R.J. Scholes

In: *Restoring Natural Capital: Views from the South*

edited by S.J. Milton and J. Aronson

Island Press: Washington, D.C.

In press (to be released in 2006)

This chapter explores the relationship between biodiversity preservation, economic development and human well-being. It uses the BII as a proxy for estimating the change in renewable natural capital in South Africa over the twentieth century, and relates it to the growth of the economy over the same period. The chapter was prepared by invitation for a book entitled *Restoring Natural Capital: Views from the South*, which forms part of the Society for Ecological Restoration International's[1] book series *Science and Practice of Ecological Restoration*. The aim of the volume is to explore the value and cost of restoring natural capital stocks, or conversely, the cost of reducing the stock of natural capital, specifically in the context of developing countries. This chapter will appear in the first section of the book, entitled '*What is natural capital and why restore it?'* It will be preceded by an introductory chapter focusing on the concepts of natural capital and its components, as well as ecosystem services; hence these concepts are not dealt with in detail in our chapter. The development of the volume included a one-week workshop in Prince Albert, South Africa, in October 2004, at which I orally presented the work in this chapter.

The chapter was co-authored by myself and R.J. Scholes. My lead role in the development of the paper is recognised by my lead authorship. I was responsible for reconstructing the historical land-use maps for each decade of the twentieth century, estimating the change in BII and relating this statistically to the gross domestic product. The authors jointly derived the uncertainties in the land use estimates over the twentieth century, and their impacts on the BII scores. I was responsible for coordinating the writing, submission and revision of the chapter. R.J. Scholes co-wrote the initial draft and provided comments on later drafts. The chapter was reviewed by an expert and the editors, and revised prior to acceptance by Island Press.

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Chapter 7: Historical changes in natural capital in South Africa: An approach using changes in biodiversity

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In the process of economic development, it is commonly observed that natural capital is partially converted to manufactured capital. We examine the conversion of natural to manufactured capital in South Africa over the twentieth century using changes in biodiversity intactness as a partial proxy for natural capital, and changes in total fixed capital stock (FCS) and gross domestic product (GDP) as indices for manufactured capital. The Biodiversity Intactness Index (BII) measures the average change in population size, relative to the pre-colonial state (pre-1700), across all species within the 'well-known biodiversity' (i.e. the plants and vertebrates) as a consequence of various types of land use. Using a historical land cover database, we estimate that the BII of South Africa has declined from 90.7% to 80.3% over the previous century. We compare this decline to inflation-corrected GDP values, which increased seventeen-fold over the twentieth century, in order to explore the changing relationship between natural capital and manufactured capital in South Africa over this period.

7.1 Introduction

The wealth of a society can be thought of as the total worth of its capital assets, comprised of manufactured, social and natural capital components. A capital asset is defined as a stock that yields a flow of goods or services (Constanza and Daly 1992; Arrow *et al.* 2003). A society's total capital assets may be broadly classified into manufactured capital (money, buildings, and other material goods), social capital (knowledge, skills and social relationships) and natural capital. Natural capital consists of non-renewable capital, such as mineral resources, and renewable capital. Renewable capital includes the resource that generates produc ts such as wood, as well as services such as erosion control that can be provided on a sustained basis if the capacity of ecosystems to deliver goods and services remains intact.

One indicator of the value of the flow of goods and services from society's capital assets is the Gross Domestic Product (GDP). GDP reflects economic flows from manufactured capital (e.g. value of products produced by a factory), and at least part of the flows from social capital (e.g. consultation fees for the knowledge and skills of a doctor) and natural capital (e.g. value of timber extracted from a forest). While the GDP does not capture all goods and services, and in particular does not account for the flows of many significant aspects of social capital (e.g. trust) or natural capital (e.g. clean air and nutrient cycling), it is the most widely used economic indicator of the productivity of society's capital base.

Sustainable development may be defined as non-decreasing social welfare, for which it has been argued the best available proxy is society's total wealth, or the value of all capital stocks (Arrow *et al.* 2003). Provided capital stocks are maintained, it is argued that the flow of goods and services on which human well-being depends can be sustained. The 'weak sustainability' criterion holds that development is sustainable provided society's total capital assets are non-declining, implying that different forms of capital are fully substitutable. The 'strong sustainability' criterion holds that there exists certain 'critical natural capital' that cannot be substituted and must be individually preserved (Ekins *et al.* 2003).

The valuation of society's manufactured capital assets, given by the Fixed Capital Stock (FCS) in a country's national accounts, is relatively well advanced in comparison to that of natural and social capital, which has only been attempted in recent years (e.g. Constanza *et al.* 1997; Arrow *et al.* 2003). Given the difficulty in valuing total capital stocks, GDP (the economic flows from society's capital stocks) is currently widely used as an indicator of the well-being of societies. The objective and measure of development is therefore commonly held to be an increasing per capita GDP, despite its shortcomings in accounting for flows from natural and social capital.

In the process of economic development, typically measured by increasing total GDP, it is commonly observed that natural capital is partially converted to manufactured capital through activities such as agriculture and mining. Flows from natural capital tend to predominate initially as societies draw on their mineral wealth, forests, wildlife and soil fertility to generate products for own consumption and trade. The revenues that result are converted, more or less efficiently, into social capital (through education and the development of institutions and services) and manufactured capital (value-adding industries and the accumulation of financial assets), which then begin to yield dividends of their own.

As flows from manufactured and social capital begin to dominate and the total wealth per capita rises substantially above the level needed to satisfy basic needs, reinvestment in natural capital may occur in certain cases through the use of a portion of a society's revenue for the protection and restoration of natural resources. This is the basis for the postulated inverse Environmental Kunznets Curve (Panayatou 1995), whereby some environmental quality indicators initially decline, but then improve again in wealthier societies. Although this pattern of development may only hold for select indicators, in cases where it does occur, the process of economic development can be thought of as taking a loan from natural capital in order to diversify the economic base, and then repaying part of that loan later. The consumptive use of non-renewable resources such as minerals cannot be 'repaid' in the same way as regeneration of renewable resources. The only way to transition to sustainability from an economy based on the extraction of non-renewable resources is to ensure that efficiency in their conversion to manufactured and social capital is high, and that the assets so created are retained within the society.

In South Africa, the reduction in non-renewable natural capital in the form of mineral reserves that occurred over the main period of mineral exploitation (1860 to present) can be estimated from mine production statistics. However, there is no general estimate of the reduction in renewable natural capital in South Africa over this period. We suggest that a proxy for at least part of the reduction in renewable capital is the change in the population sizes of various forms of wild organisms (biodiversity) that has occurred over the colonial and post-colonial period. The populations of different wild organisms can be seen as an expression of natural capital, since most ecosystem goods and services rely on specific organisms for their production, and the size of the flow depends on the abundance of the organisms.

Scholes and Biggs (2005) developed the 'Biodiversity Intactness Index' (BII) to estimate changes in the mean population sizes of a wide range of organisms when subjected to a range of different land uses. Applying the approach to South Africa in year 2000, they estimated an overall reduction of about 20% in the population sizes of plants and vertebrates relative to their pre-colonial levels, with much higher reductions in certain groups such as the large mammals. The

BII is conceptually and mathematically equivalent to the Natural Capital Index (Ten Brink *et al.* 2002) which has been applied in several European countries.

The objectives of this chapter are firstly to apply the BII to changes in land use in South Africa between 1900 and 2000, in order to estimate the rate and pattern of loss of renewable natural capital. Secondly, the changes in BII are compared with changes in GDP and FCS over the same period in order to describe the process of conversion between natural and manufactured capital.

7.2 Estimating historical changes in biodiversity intactness

The method for calculating the BII is detailed in Scholes and Biggs (2005). Briefly, BII is the average, across all species of well-known terrestrial biodiversity (plants, mammals, birds, reptiles and amphibians), of the populations at a particular time relative to the populations at a reference time. The reference in this study was nominally the pre-colonial period (pre-1700). In practice, contemporary abundances in large protected areas were used as a proxy for the pre-colonial state. Changes in population abundances were estimated by a panel of sixteen experts, for a range of land uses of increasing intensity (Table 7.1). Estimates were made for each of about ten functional types per broad taxon, for each of six major biome types (forest, savanna, grassland, arid shrublands, *fynbos* and wetland) in southern Africa, and aggregated up to the taxon level by weighting by the species richness of each functional type. Weighting these relative population abundance estimates for each taxon in each land use by the area affected and the number of species (richness) in the taxon gives the BII:

$$BII = \frac{\sum_i \sum_j \sum_k \mathrm{R_{ij} A_{jk} I_{ijk}}}{\sum_i \sum_j \sum_k \mathrm{R_{ij} A_{jk}}}$$

where

I_{ijk} is the population of taxon i under land use activity k in ecosystem j, relative to a reference population in the same ecosystem type;

R_{ij} is the richness (number of species) of taxon i in ecosystem j; and

A_{jk} is the area of land use k in ecosystem j.

Table 7.1 Land uses classes and associated data sources used for the estimation of the biodiversity intactness index.

Land use class	Description	Examples	Data source
Protected	Minimal recent human impact on structure, composition or function of the ecosystem. Biotic populations inferred to be near their potential.	Large protected areas, 'wilderness' areas.	World Database on Protected Areas (IUCN & UNEP 2003). All designated protected areas of IUCN categories I-V.
Moderate use	Some extractive use of populations and associated disturbance, but not enough to cause continuing or irreversible declines in populations. Processes, communities and populations largely intact.	Grassland and savanna areas grazed within their sustainable carrying capacity.	All remaining areas not classified into one of the other five categories.
Degraded	High extractive use and widespread disturbance, typically associated with large human populations in rural areas. Productive capacity reduced to approximately 60% of 'natural' state.	Areas subject to intense grazing, harvesting, hunting or fishing, areas invaded by alien vegetation.	1995 South African National Land Cover map (Fairbanks *et al.* 2000).
Cultivated	Land cover permanently replaced by planted crops. Most processes persist, but are significantly disrupted by ploughing and harvesting activities. Residual biodiversity persists in the landscape, mainly in set-asides and in strips between fields (matrix), assumed to constitute approximately 20% of class.	Commercial and subsistence crop agriculture.	1995 South African National Land Cover map (Fairbanks *et al.* 2000).
Plantation	Land cover permanently replaced by timber plantations. Matrix areas assumed to constitute approximately 25% of class.	Plantation forestry, typically pine and eucalypt species.	1995 South African National Land Cover map (Fairbanks *et al.* 2000).
Urban	Land cover replaced by hard surfaces such as roads and buildings. Dense populations of people. Most processes are highly modified. Matrix assumed to constitute 10% of class.	Dense urban and industrial areas, mines and quarries.	1995 South African National Land Cover map (Fairbanks *et al.* 2000).

Changes in BII over the twentieth century were assumed to be a function only of changes in the extent of different land uses (A_{jk}). Changes in the population impacts of different land uses (I_{ijk}) and the potential species richness of different biomes (R_{ij}) were assumed to be negligible over the time period examined. Biome boundaries in South Africa were assumed to have remained unchanged over the twentieth century.

The spatial pattern of the land uses listed in Table 7.1 between 1900 and 2000 was reconstructed per biome, at the magisterial district level, at a decadal timestep. Data on the progression of cultivated area during the twentieth century were obtained from agricultural statistics collated at the magisterial district level by Biggs and Scholes (2002). Where a district spanned more than one biome, the area under cultivation in each biome was allocated in proportion to the cultivated area in the 1995 Land Cover map of South Africa (Fairbanks *et al.* 2000) (Figure 7.1). The change in afforested area over the twentieth century was obtained in the same manner. The location and dates of establishment of designated protected areas of IUCN categories I to V were obtained from the World Database on Protected Areas (IUCN & UNEP 2003). The urban land area in each magisterial district was calculated from 1995 Land Cover map and back-calculated for each decade of the twentieth century by adjusting the 1995 area in proportion to the population which was urbanized in each province by the start of each decade compared to the percentage urban population in 2000. Official population census data were obtained from Statistics South Africa. Land not in the protected, cultivated, plantation, urban or degraded categories was assigned to the moderate use class.

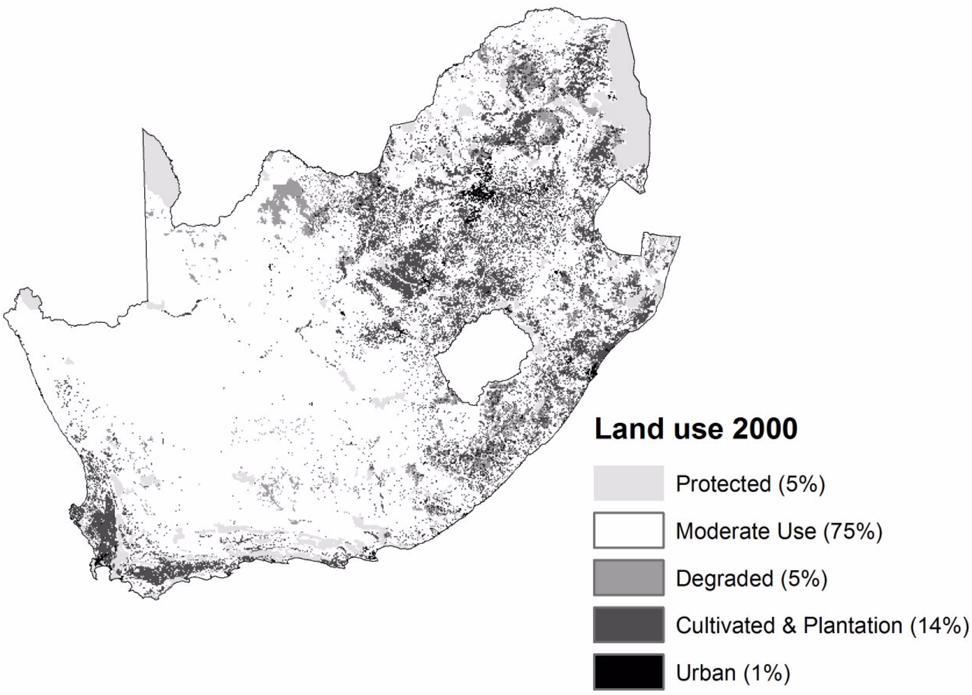

Figure 7.1 The distribution of land uses *circa* 2000, in the six categories used for the calculation of BII (see Table 7.1). The basic map is derived by lumping land cover classes in the South African National Land Cover Map (Fairbanks *et al.* 2000), and combining this with the protected areas of IUCN categories I to V from the World Database on Protected Areas (IUCN & UNEP 2003).

The area of degraded land was reconstructed by building a narrative of the major changes and their causes, location and timing during the twentieth century. The formalization of land tenure restrictions, the increase in domestic livestock numbers, and the occurrence of severe droughts and periods of economic depression were the key considerations (Beinhart 1984; Hoffman 1997; Hoffman & Ashwell 2001). Based on the historical narrative, time-courses for these drivers, and the consequent extent of degradation relative to 1995 were derived for 17 different regions of the country. These 'degradation sequences' were applied retrospectively to the 1995 distribution of degraded land in the South African Land Cover map in the relevant regions to derive the spatial distribution of degradation for each decade of the twentieth century.

The estimated fractions of South Africa in the various biodiversity impact classes are given per decade in Table 7.2. Our uncertainty regarding the estimates of BII over the twentieth century are related to the level of uncertainty in the estimates of land use distribution over time. Cultivation has the largest impact on the estimate of BII, since the area involved was about 12% of South Africa in the year 2000, and the mean impact score was 0.25, i.e. a 75% reduction in biotic populations. Cultivation was responsible for half the reduction in BII by the year 2000. The data on cultivated area (and plantations) are the most reliable in the set, with an error of no more than 1% absolute, which translates to a maximum absolute error of 0.7% in the BII score for the year 2000.

Table 7.2 Estimated percentages of South Africa in six land use categories, per decade over the twentieth century. The total land area of the country is 1.2 million km^2.

Year	Protected	Moderate use	Degraded	Cultivated	Urban	Plantation
1900	0.00	95.94	0.41	3.37	0.26	0.02
1910	0.32	94.65	0.62	3.97	0.34	0.10
1920	0.32	94.02	0.85	4.29	0.37	0.16
1930	1.88	90.51	1.42	5.52	0.42	0.25
1940	2.70	87.87	2.06	6.56	0.50	0.31
1950	2.72	86.39	2.41	7.52	0.60	0.37
1960	2.79	82.80	2.68	10.37	0.67	0.69
1970	3.02	80.52	3.81	11.00	0.83	0.81
1980	4.03	78.76	4.26	11.14	0.87	0.94
1990	4.39	76.55	4.64	12.33	1.08	1.01
2000	5.30	74.76	4.95	12.23	1.29	1.47

The second largest effect, accounting for a quarter of the reduction in BII, is due to moderate extractive uses, such as cattle ranching. Although the mean impact is low (0.93), an extensive area is affected (75% of South Africa in the year 2000). Since the area under extensive 'moderate use' is derived by difference (Table 7.1), the uncertainty in the total area classified as such depends on the accuracy with which the other major classes are mapped. Cultivated area has been well mapped; similarly, the protected area is reliably mapped and dated, at least for the major areas.

The method used for mapping the development of urban areas was crude, but the total area is small (1.3%), so even with a mean impact score of 0.13, the potential error is limited to less than a tenth of the observed reduction of BII. The largest amount of uncertainty is associated with the 'degraded' class, which occupied about 5% of South Africa in 2000, and results in nearly a 50% reduction in population abundances where it occurs. Degradation is responsible for about an eighth of the total biodiversity impact; if the uncertainty in the area classified as degraded is $\pm$2% absolute, then the absolute error in BII is 0.9% from this source. The total error in the estimate of BII resulting from the uncertainty in the areas under different land uses is therefore less than 2% absolute.

7.3 The conversion of natural capital into manufactured capital

Taking the BII as a partial proxy for natural capital, changes in BII, FCS and GDP over the twentieth century support the conceptual model of a partial conversion of natural capital into manufactured capital during the economic development of South Africa (Figure 7.2). For the first two centuries following the arrival of European settlers in South Africa in the seventeenth century, economic activity was based almost entirely on the use of renewable natural resources: grazing land, cultivating crops, hunting wildlife and cutting timber from forests. The total population grew slowly, and the impact on the natural resource base was spatially relatively limited. By 1900, the fertile valley soils of the south-western Cape were mostly converted to crop agriculture; the very limited extent of indigenous tall forest around Knysna was approximately halved by unsustainable logging and fires; and the large herds of antelope that grazed the interior plateau had been decimated (Hoffman 1997). The concentration of these impacts in the highly plant diverse *fynbos* region and the lack of protection for indigenous mammal species (Carruthers 1995) accounts for the 10% average reduction in the populations of South Africa's flora and fauna that had occurred by the start of the twentieth century.

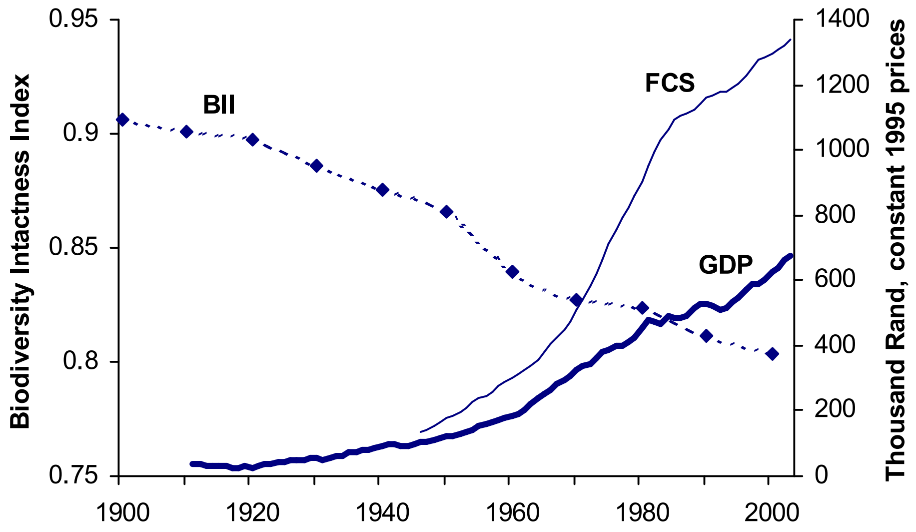

Figure 7.2 Changes in the biodiversity intactness index (BII), total fixed capital stock (FCS) and gross domestic product (GDP) of South Africa over the period 1900 to 2000. Data on total fixed capital stock and GDP were obtained from the South African Reserve Bank. Data were not converted to US dollars due to the absence of well operating financial markets in the early 190 0s. For indicative purposes only: 1 USD = R3.65 in mid -1995.

The discovery of diamonds in 1867, and large deposits of gold in 1886 (Byrnes 1996), triggered the switch to the dominance by manufactured capital that took place over a period of about a century. Mineral discoveries attracted large numbers of European immigrants and African migrants, and led to a rapidly expanding urban population. This nec essitated the establishment of surplus-producing commercial agriculture to provide food, as well as commercial forest plantations to provide support timber for the mines. A growing tax base and strengthening government institutions, coupled with dramatic t echnological advances globally in the fields of sanitation and medicine, led to a steady reduction in mortality rates in both African and European populations; consequently, that the total population grew rapidly throughout the twentieth century (SSA 2004). The increased population, abetted by expanding markets and technologies, including newly imported breeds of livestock, domestication of the ostrich, farm mechanization, irrigation, new maize cultiva rs and agricultural chemicals, began to transform the landscape to an unprecedented degree and extent (Hoffman 1997). Between 1920 and 1960, aided by government loans and subsidies, there was a steady expansion in the area under cultivation, particularly in the interior high altitude grassland region of South Africa (Biggs and Scholes 2002), and a corresponding decrease in BII.

Degradation of land in areas under moderate extractive uses, such as cattle ranching, has been another significant factor leading to a loss of wildlife populations in South Africa. Conflict over land and political domination by the European population led to the African population being restricted to about an eighth of South Africa by the Land Acts of 1913 and 1936. African urbanization was initially blocked by legislation, resulting in excessive pressure on natural resources in their designated 'homeland' areas (Biggs and Scholes *in press*; Beinhart 1984; Hoffman & Ashwell 2001; Nieuwoudt & Groenewald 2003): population densities in these areas were five to ten times greater than those of the rural areas in the rest of South Africa (SSA 2004). The average household income was also substantially lower, and lack of capital for agricultural inputs and conservation, coupled with the small landholdings, insecure tenure and lack of technology, advice and markets, led to deteriorating land productivity in many of these areas (Hoffman & Ashwell 2001). The greatest increase in degraded area since 1900 occurred during the decade of the 1960s (Table 7.2), largely related to the forced removal of thousands of Africans and their resettlement in the homeland areas during this time. Based on the 1995 land cover map (Fairbanks *et al.* 2000), 73% of the degraded land in South Africa was located in the former homeland areas, which constituted only 14% of South Africa's land surface. Nevertheless, land degradation was also an issue of significant concern in white-owned commercial farming areas during the twentieth century. Widespread farming failures, triggered by droughts and economic depression in the early 1930s led to substantial increases in the degraded area during this decade.

The pattern of decline of BII during the twentieth century was unsteady, largely tracking the progression of change in cultivated area. The major loss occurred during the 1950s (absolute decrease of 2.6%), coupled primarily to the 3.2% absolute increase in the area under cultivation and afforestation (Table 7.2). The next highest loss occurred in the decade of the 1960s, related to the large increase in the estimated degraded area during that decade. The 1970s, on the other hand, saw the smallest decline in BII of any decade since 1900. During this decade, there was a withdrawal of many of the agricultural subsidies that had encouraged the expansion of cultivated land (which had in any case already occupied the most favorable locations), resulting in a leveling off of the total cultivated area (Biggs and Scholes 2002). The seventies was also a decade of good rainfall (Preston-Whyte & Tyson 1988), and thus slower land degradation. The slow but continuing decrease in BII during the 1980s and 1990s is associated with marginal increases in the area under cultivation, and degradation as a result of drought and continued high population pressure in many of the former homeland areas, which are still poor and heavily reliant on ecosystems for products such as fuelwood.

Growth in the GDP broadly parallels the growing population, but shows spurts in the early 1920s and from 1946 to 1970, corresponding to global economic growth in the post-war periods, and its effects on demand for the mineral, manufactured and agricultural goods produced by South Africa. Encouraged by the alternative livelihood opportunities presented by the burgeoning economy (Byrnes 1996), nearly 80% of the European population had become urbanized by 1950 (SSA 2004), and were mainly employed in the manufacturing and services sectors. The rapid growth in GDP between 1960 and 1980 was derived mainly from these sectors, rather than from renewable natural capital, and reinvested in FCS, in the form of infrastructure (Figure 7.2). Fluctuations in GDP during the 1980s were the result of a severe drought between 1981 and 1985 (Preston-Whyte & Tyson 1988), large fluctuations in the export price of gold (on which South Africa was heavily reliant), and rising security costs and labor disputes associated with the long-term effects of political suppression (Byrnes 1996). The end of apartheid in 1994 and improved macro-economic management brought a return of foreign investment and rapid growth of the economy. By 2000, the majority of Africans were also urban dwellers employed in secondary and tertiary sectors of the economy, which accounted for 90% of GDP (SARB 2004).

Plotting the pairs of BII and GDP numbers for each decade (Figure 7.3) suggests an inverse exponential relationship between natural capital and total flows of goods and services. Initially, the decline in BII per unit increase in GDP is rapid, as society's revenue is mainly accrued from natural resources, and particularly from the transformation of the landscape into agricultural land uses. These dividends are reinvested in the establishment of the manufacturing and service sectors, which in time begin to yield dividends of their own, so that further increases in GDP accrue less and less from natural capital stocks and increasingly from manufactured and social capital. The growth of GDP therefore becomes increasingly decoupled from the reduction in natural capital. Although it could not be tested here due to the lack of FCS data prior to 1946, given the high linear correlation between GDP and FCS ($r^2 = 0.981$, $p < 0.001$, $n=58$), it is likely that a similar relationship holds between natural capital and manufactured capital.

In our opinion, it is likely that the BII will decline at a slow rate in South Africa in the next few decades. Apart from the decoupling mechanism described above, large areas of South Africa are not amenable to extreme land transformation: they are too steep, too infertile, too dry or too remote for intensive land uses such as cultivation or urbanization. However, they are not immune to other forms of degradation, such as tree-clearing, overgrazing and alien plant invasion. The principle challenge for biodiversity conservation in the medium term is thus preventing the extensive areas currently under 'moderate use' from becoming degraded. While not apparent in the data for South Africa at this stage, the current reinvestment of manufactured capital into a rapidly expanding nature-based tourism sector with the associated increase in privately-managed protected areas and indigenous mammal populations (Scholes & Biggs 2004) may conceivably lead to a small increase in BII over the first decade of the twenty-first century.

Three significant concerns remain for the protection of biodiversity. The first is climate change, which has the potential to have a major effect on South African biodiversity, particularly in a fragmented landscape. The second is the widespread effect of industrial and urban pollution (particularly the deposition of nitrogen on land, and acid drainage in rivers) in an increasingly urban and industrial future. The third is the insidious effect of habitat fragmentation on population viability, which is not well accounted for in the BII approach.

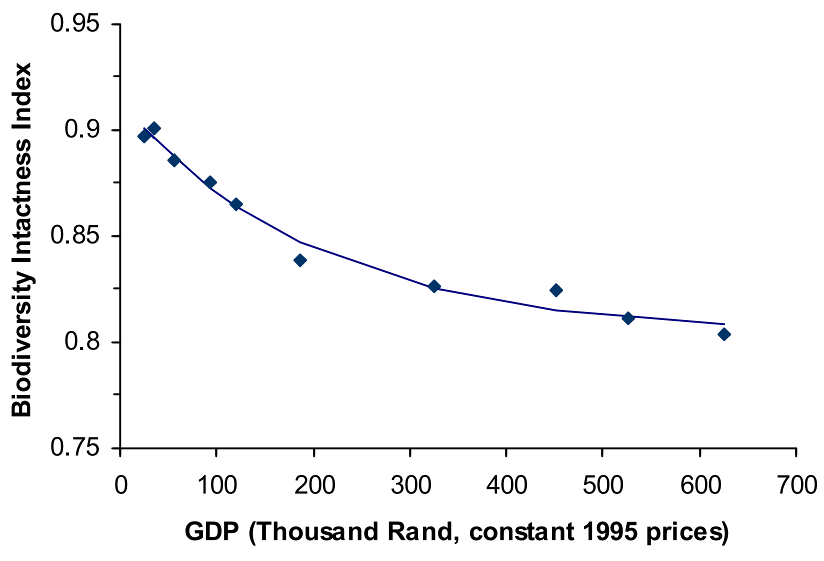

Figure 7.3 The relationship between BII and GDP for South Africa. Data points are for the end of each decade between 1900 and 2000. The equation of the fitted regression line is BII = 0.804 + 0.110 × 0.999994973 ^{GDP} (r^2 = 0.976, $p < 0.001$, n = 8). This form was empirically selected and should not be extrapolated.

7.4 Conclusion

Taking the Biodiversity Intactness Index (BII) as a partial proxy for natural capital, and Fixed Capital Stock (FCS) and Gross Domestic Product (GDP) as indices for manufactured capital, the changes that occurred over the twentieth century reflect a partial conversion of natural capital into manufactured capital. Using a historical land cover database for South Africa, we estimate that the BII has declined from 90.7% to 80.3% over the previous century, i.e. by the year 2000 there had been an average reduction of 20% in the population abundances of all indigenous plant and vertebrate species. Over the same period, the area under cultivation increased by an area of 110,000 km^2 (from 3% to 12% of South Africa's land area), while degraded and protected areas have grown from nearly nothing to approximately 70,000 km^2 (5% of the land area) each. The vast majority of South Africa (95% of the land area in 1900, 75% in 2000) is under 'moderate extractive use', mainly for livestock ranching, and increasingly for nature-based tourism ventures.

The pattern of decline in BII over the twentieth century was unsteady, and largely reflects the relative change in the major land uses. BII declined at a relative rate of 0.12% per year over the past century; this compares with an average population growth rate of 2.3% per year, and an average GDP growth rate of 4% per year. The greatest loss in BII occurred during the decade of the 1950s (2.6% absolute), related to a large increase (3.2% absolute) in the area under cultivation and afforestation; the smallest loss occurred during the decade of the 1970s (0.3% absolute), related to a leveling-off of the area under cultivation.

Inflation-corrected GDP increased seventeen-fold over the twentieth century. Until the mid-twentieth century, GDP increased mainly by drawing on natural capital stocks, particularly through mining and conversion of land to agriculture. From the 1950s onward, the growth in the economy of South Africa has been increasingly based on the manufacturing and service sectors (SARB 2004), and the rate of decline in BII has decreased. A plot of the relationship between BII and GDP suggests that during the economic development of a country such as South Africa, the economy initially draws on natural capital stocks, but the dividends are then reinvested in manufactured and social capital. Over time, further growth in GDP accrues increasingly from manufactured and social capital, and becomes increasingly decoupled from a reduction in natural capital.

7.5 Acknowledgements

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Chapter 8: Conclusion

Chapter 8: Conclusion

This study set out to derive a macro-ecological indicator for assessing the state of biodiversity, and to test it by application in southern Africa. The original indicator developed in this study is called the Biodiversity Intactness Index (BII). This chapter evaluates the BII in relation to the criteria that such a macro-ecological indicator were intended to satisfy. The condition of biodiversity in southern Africa, its historic changes in South Africa, its possible future trajectories in the region, and the implications for biodiversity management are summarized in section 8.2. The final section suggests further avenues for research.

8.1 Evaluation of the BII as a macro-ecological indicator

The BII is a promising candidate for a synthetic macro-ecological indicator of biodiversity status when assessed against the criteria set by the Convention on Biological Diversity (CBD 2003a; CBD 2003b; CBD 2003c), which were expanded on in the objectives of this study (see section 1.3) and in the review of other candidate indicators in Chapter 2. Based on this study, I conclude that the BII is scientifically sound. It represents an actual physical quantity, namely the average remaining populations of a wide range of organisms relative to their reference populations, which synthesises the impact of many factors on biodiversity. Unlike extinction- or richness-based indicators, the BII is sensitive to the process of biodiversity loss in its early phases, which typically manifest as a gradual reduction in the *abundance* of populations of affected species, and may eventually end in the extinction of some. Furthermore, the BII takes account of the graduated nature of the impact of different human activities. As shown in chapter 4, the BII is sensitive the most important driver of biodiversity loss at a global scale and in southern Africa, namely land use change (MA 2005). At the same time, it is not overly sensitive to reasonable differences in the interpretation of broad land use classes. It also takes account of other factors that can be expressed on an area-affected basis, such as harvest pressure and pollution. Chapter 4 establishes the scientific soundness of applying the BII at widely differing spatial scales and directly comparing results between scales. The uncertainty in the estimate of BII is quantifiable, and can be used to direct further research effort.

The BII is highly relevant to policy. This is illustrated in Chapter 3, where the BII is applied to southern Africa, and motivated as a possible tool for evaluating progress towards the globally-adopted 2010 target of reducing the rate of biodiversity loss (UN 2002). The BII synthesizes a large volume of data, covering land cover, ecosystem extent, species richness and population abundance, to provide a high-level overview for the public and politicians, which is intuitively understandable, easy to communicate, and meaningful to policy-making. The analysis presented in Chapter 3 formed the core of the Southern African Millennium Ecosystem Assessment (SAfMA) regional-scale assessment of the condition and trend of biodiversity (Scholes & Biggs 2004). The BII can be used to compare a baseline situation to a policy-target (as illustrated in Chapter 3), to explore scenarios of the future (as illustrated in Chapter 6), and to reconstruct historical changes (as illustrated in Chapter 7). Chapter 7 also shows that the BII can be related to concepts such as economic value and human well-being, in order to evaluate trade-offs, which lie at the core of policy- and decision-making. The BII is amenable both to high-level aggregation, for example to the national or regional level, and to disaggregation in several ways. It can be downscaled spatially as far as the fundamental resolution of the base data will permit. It can be disaggregated by political or ecosystem boundaries, by taxonomic or functional group, or by human activity causing biodiversity loss. This allows the BII to meet the needs of users at multiple levels and different spheres of decision-making, as illustrated in Chapters 3 and 5.

The BII is practical and affordable to implement. The BII can be calculated using a variety of data sources, as well as hybrids of high-resolution and low-resolution data, and the results are convergent (illustrated in Chapters 3 and 4). This makes it possible to obtain initial estimates relatively quickly, and then improve them incrementally. In particular, making use of expert-derived data on the population impacts of various land use activities means that the BII can be estimated with several weeks of effort, rather than the decades needed for detailed population surveys. In localities where detailed data are available, these can be used in place of or together with expert-derived data. The BII can therefore complement data-intensive indices such as the mathematically-similar Natural Capital Index that has been implemented in the Netherlands (Ten Brink *et al.* 2002).

No single biodiversity index is sufficient for all purposes. The BII provides an overview by averaging across species; it is not intended to highlight individual species that are under threat, and should be used to complement indicators such as the Red List of Threatened Species (IUCN 2002). The principle disadvantage of the BII as formulated and implemented in this study is that it may be insensitive to gradual processes, such as the long-term effects of habitat fragmentation, pollution and climate change, and the knock-on effects of local species extinctions. Conceptually, these factors should be included in the population-impact factors related to specific land uses or areas, but in practice field experts are unlikely to account for these in their estimates. For calculations spanning more than a century forwards or backwards in time, it is therefore strongly suggested that adjustments be made to the richness-weighting and population impact factors in order to account for the effects of such processes.

8.2 Key findings: The biodiversity intactness of southern Africa

Application of the BII to southern Africa indicates that the condition of biodiversity in the region is better than we sometimes fear. Averaged across the well-described form of biodiversity, i.e. the plants and vertebrates, it is estimated that 84 ± 7% of the pre-colonial number of wild organisms persist in present-day southern Africa, despite greatly increased human demands on ecosystems over the past 300 years. This is substantially greater than that suggested by simply considering the formally protected area (only 8% of southern Africa's land surface) (IUCN & UNEP 2003), and points to the importance of areas *outside* reserves in the preservation of biodiversity. This study quantifies the role of these areas: 85% of the remaining wild organisms in southern Africa are estimated to be in the extensive areas under moderate extractive uses, which constitute 78% of southern Africa's land area. Given the practical limits to expansion of the area under formal protection, this suggests that the policy action with the greatest potential to prevent further loss of biodiversity is to prevent land currently under moderate extractive use from becoming degraded. While cultivation and urbanisation have a higher impact on the populations of wild organisms per unit land area than does degradation, a much smaller fraction of the land is at risk. This holds under a range of feasible socio-economic scenarios for the 21st century.

The impact of humans on biodiversity is expressed very selectively. With a BII score of 71%, mammals are the most impacted taxonomic group. Species that are large-bodied, and thus easy to hunt or harvest, are the most affected. This is especially so if they are either valuable, or in direct conflict with aspects of human well-being, such as large predators or large herbivores that compete with domestic ungulates. While the threat to these species should not be discounted, they constitute only a small fraction of the total biodiversity. The vast majority of species are affected mainly through loss of habitat to cultivated lands or urban areas. Accordingly, the greatest impact on biodiversity has occurred in the grassland biome, followed closely by the *fynbos*. Aggregated at a national level, the most densely populated countries in southern Africa, Lesotho and Swaziland, are most highly impacted, with BII scores of 69% and 72% respectively.

The rate of change of BII in southern Africa over the decade of the 1990s is estimated as an absolute decline of 0.8%. In order to meet the 2010 target, the decadal rate of change in BII, estimated in a consistent manner, would need to be lower than 0.8% when calculated for the first decade of the 21st century. A scenario analysis, incorporating a range of economic and demographic conditions, shows that the rate of decline in BII in southern Africa is unlikely to decrease significantly until the late 21st century. Given the need for economic development to meet poverty reduction targets in the region, some drawing down of natural capital in the form of biodiversity may be a reasonable policy decision. However, specific attention needs to be directed at ensuring that natural capital is efficiently converted into other forms of societal wealth, and not simply consumed or exported.

The relationship between BII and Gross Domestic Product (GDP) in South Africa over the 20th century suggests that economic development initially drew heavily on the natural resources, but has become progressively decoupled from the loss of biodiversity. South Africa has experienced an absolute decline of 10.3% in the value of BII over the 20th century, to reach a score of 80% in the year 2000, while the GDP has increased eighteen-fold over the same period. The greatest loss in BII occurred in the decade of the 1950s, related to a large increase in the area under cultivation and afforestation. The land area suitable for cultivation appears to be the primary determinant of the level at which the rate of decline in BII has currently slowed.

8.3 Further research and applications

The development and application of the BII suggests several areas that need further research in order to better understand the effect of human activities on biodiversity and its rate of loss. The most important of these are the quantification of the impact of various land use activities on the abundance of populations of different species. To date, much of the conservation effort has been focused on protected areas and threatened species, and less attention has been paid to the importance of areas outside reserves in biodiversity conservation. Where work has been done on populations outside protected areas in southern Africa, it has been heavily biased toward specific land cover types, such as plantations and degraded rangelands. For most taxonomic groups, almost no population studies have been done in cultivated or urban landscapes. Research into population abundances outside reserves may well stimulate investigations into how specific land uses, such as urbanisation, can be managed to reduce their impacts on biodiversity.

More accurate data on the extent and nature of different types of *land-use* (not simply satellite-derived *land-cover*) will help to better refine estimates of the extent and nature of human impacts on biodiversity. In comparison to our knowledge on land-use extents and impacts, our knowledge on the historical spatial distribution of species is relatively well advanced.

The BII lends itself to application in other parts of the world, or even to a global analysis. Preliminary comparisons of our expert-derived population impact estimates to field data in other parts of the world suggest that, at least at this broad level of land use classification, our estimates agree well with such data. However, if the analysis were to be extended, this correspondence would need to be formally verified, and at a minimum our expert-derived estimates would need to be supplemented with additional estimates for biomes and land use types not present in southern Africa. The issue of comparable baselines would also need to be addressed, given that areas such as Europe were already extensively transformed by the start of the modern era. Ways of estimating the impact on population abundance of processes such as climate change, habitat fragmentation and chronic, low-level pollution represent additional research challenges.

The BII could be extended to apply to marine ecosystems, and adapted for application in ecosystems that are essentially linear rather than area-based, such as coastal and freshwater systems. This would require a trivial change in the algorithm (substitution of length for area). Species richness data for aquatic plants and vertebrates exist. The major task would be to define impact categories analogous to those used for the terrestrial BII calculation, and to collect expert judgements (and/or published field data) regarding the impacts of these categories on the populations of various functional groups involved.

The 2010 target of reducing the rate of biodiversity loss at global, regional and national levels needs to be evaluated in the context of the Millennium Development Goals[1]. A better understanding of the trade-offs between biodiversity as a form of natural capital, and other

[1] www.developmentgoals.org

forms of societal capital needs to be developed. A more nuanced target may be needed, which accounts for the different stages of economic development between developed and developing countries. In addition to setting targets for the rate and absolute amount of biodiversity loss (which is essential), it would be prudent to relate these to growth targets in other forms of societal capital in order to ensure sustained net total capital in society (sustainable development). Understanding the trade-offs and the process of conversion between natural capital and other forms of capital requires further development and regional parameterisation of large integrated models which link the environment, the global economy, technology and societal attitudes and organization.

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Appendices

Appendix 1: The BII method

The Biodiversity Intactness Index (BII) gives the average population size of a wide range of organisms, relative to their selected baseline populations in a particular area. The BII is calculated from three basic input factors: Richness (R_{ij}), area (A_{jk}) and relative population size (I_{ijk}). These factors are defined in terms of specific taxa (i), ecosystems (j) and land uses (k). This appendix details the considerations in defining i, j and k, and for determining R_{ij}, A_{jk} and I_{ijk}.

The BII can be calculated either on a raster (pixel) basis or a vector (polygonal) basis, and the results are equivalent. The choice of method will depend primarily on the form of the underlying data, and the type of analytical software available.

Taxa (i)

It is recommended that the BII be calculated across all indigenous species within the broad taxonomic groups that are reasonably well described. In the southern African case, this included plants and vertebrates (mammals, birds, reptiles and amphibia). Although diverse, invertebrates and microbes were excluded as they are poorly documented. If BII is to be compared between different regions, the same broad taxonomic groups should be covered in each of the regions.

Alien species were excluded from the BII calculations for southern Africa. The pre-modern baseline populations for these species are zero, which creates both mathematical and interpretive difficulties as any comparison of current population levels to the baseline results in (infinitely) large values for I_{ijk}. However, under certain circumstances alien species may be deemed 'naturalized', and it may make sense to include them in the estimation of BII. In such cases, the baseline populations for these species should not be set at zero, but at the stabilized 'naturalized' population size, or some non-zero targeted population size.

Land uses (k)

Land uses are defined by the major human activities impacting on biodiversity in a particular region. All land use activities are expressed on the basis of the area affected. It is suggested that the number of classes be limited to less than ten, in order to keep the number if I_{ijk} estimates required manageable. For the southern African case, six categories were defined: protected, moderate use, degraded, cultivated, plantations and urban (see Scholes & Biggs 2005 for definitions). These form a rough continuum of increasing degrees of impact. A land use map for southern Africa was derived from land tenure boundaries and several sources of land cover data. Where classes from different sources overlapped, the highest impact land use class was assigned. It is important to define the resolution of the land use map (or the various land use classes, if their resolutions differ) as it affects the I_{ijk} estimates (see discussion below).

Impacts such as fragmentation or pollution are incorporated in the definition of the land use categories. In the southern African example, typical urban pollution impacts were explicitly taken into account in the estimates of I_{ijk} for urban areas. Although not done in the southern African case, where pollution or fragmentation impacts cannot be associated with an entire class, multiple classes can be defined, e.g. large protected areas, and small protected areas (to account for fragmentation impacts). Separate I_{ijk} estimates would then be collected for each class.

Ecosystems (j)

Broad-scale associations of flora and fauna can be defined as ecosystems, and can typically be arranged into several hierarchical levels. Spatial data on the extent of ecosystem types are needed to define the area to which specific I_{ijk} estimates apply, and to attach the appropriate richness weightings (R_{ij}) to each of these areas. Note that the data for R_{ij}, A_{jk} and I_{ijk} can be at different hierarchical levels. In the southern African case, the WWF Ecoregions (Olson *et al.* 2001) were used, and aggregated into six high-level biomes (forest, savanna, grassland, shrubland, *fynbos* and wetland). R_{ij} and A_{jk} were determined in terms of the disagreggated WWF ecoregion data, while I_{ijk} estimates were collected at the biome level, and then associated with each of the ecoregions.

Richness (R_{ij}) and Area (A_{jk})

Species richness data is typically available as total species counts for each broad taxonomic group, per ecosystem type. In southern Africa, species richness data associated with the WWF Ecoregions were used. Where available, the potential geographical distributions for individual species can be used. Individual species distribution data can be used in conjunction with ecosystem-level data for species for which more detailed data is unavailable. When using individual species distribution data, it is more practical to calculate the BII on a raster (pixel) basis, rather than a vector (polygonal) basis.

The area of a particular land use within a specific ecosystem type (A_{jk}) is simply determined by overlaying ecosystem and land use maps. In the southern African case, the land use map that had been derived for the purpose of calculating the BII was overlaid on the WWF Ecoregion map.

Relative population size (I_{ijk})

The first step in estimating I_{ijk} is to define a meaningful and practical reference or baseline population. In the southern African context, pre-modern populations (pre-1700) in different ecosystem types, unimpacted by modern-era land uses, were used as a conceptual baseline. For the majority of species, populations in large protected areas served as a practical proxy for this state (i.e. I_{ijk}=1, where k = protected). Only for select species with large home ranges were large protected areas impacted compared to the conceptual baseline, in the judgment of some experts. In parts of the world that had already been highly transformed by the beginning of the modern period, alternative reference points can be defined, such as a specific baseline year within record or reliable memory. The same baseline should be used for comparisons between BII at different points in time or between different regions.

Where available, field data can be used to estimate I_{ijk}, the population size of species group *i* under land use activity *k* in ecosystem *j* at a specific point in time, relative to a baseline population in the same ecosystem type. In most cases, field data will be available only for a few species in a few locations. Where a great deal is known about the underlying drivers, population models could be used to generate the I_{ijk} matrix. In the southern African example, the I_{ijk} matrix was generated using expert judgement, and validated against available field data. Multiple data sources can be used to estimate I_{ijk}, such that the best-available data are used for each species.

To assist the expert interview process conducted in southern Africa, each broad taxonomic group was subdivided into five to ten functional types. These functional types were defined as groups of species that respond similarly to the set of land uses under consideration. The exact number of types defined is a practical, rather than theoretical, issue. Across all taxonomic groups the functional types were defined primarily by body size, trophic niche and reproductive strategy. Thus for plants, a variant of the Raunkier classification worked well, and for birds and mammals a division of 'small', 'medium' and 'large' by 'herbivore', 'carnivore' and 'omnivore' was successful. On the other hands, all frogs were small insectivores, so breeding strategy was used as a functional classifier.

Each expert was asked to estimate the impact of each land use activity within the context of each broad ecosystem type (biomes, in the southern African example) on each functional type in their speciality taxonomic group (different experts were interviewed for each broad taxonomic group). The southern African results suggest that land use (*k*) is the overriding factor impacting on the estimates of I_{ijk}. In comparison, the differences in the impact of specific land uses between ecosystem types, is relatively small (e.g. the impact of cultivation in savanna as opposed to cultivation in grassland areas, is small compared to the difference in impact between cultivation and urbanisation). Given the coarseness of our current knowledge regarding I_{ijk} and the practical constraints of collecting or generating this data, it therefore appears to be sufficient to define I_{ijk} at the biome-level in the first instance. Where finer level ecosystem data are available for R_{ij} and A_{jk}, it is recommended that these be used in calculating BII.

It is important to thoroughly discuss the land use categories with the experts *before* beginning the judgment exercise, so that they are conceptually 'calibrated' and can visualise the state that has been defined. To assist the experts, a local example of an area classified under each land use type in each biome was given, e.g. Kruger National Park is an example of a protected area in a savanna, and Johannesburg is an example of an urban area in a grassland region. It is also important that the experts take account of the resolution of the land use classification in making their estimates. In regional to global studies, land cover is typically mapped at a resolution of about 1 x 1 km, and an area classified, as for example, 'cultivated' almost always has inclusions of uncultivated land in set-asides and strips between fields (referred to as the residual matrix). In the southern African case, it was assumed that, mapped at a 1 x 1 km resolution, the matrix constitutes approximately 20% of the cultivated class, 25% of the plantation class, and 10% of the urban class.

Estimates of I_{ijk} were aggregated up from the functional type level to the broad taxonomic level by weighting the estimates for each functional type (*f*) by the number of species in that group in the particular ecosystem type ($I_{ijk} = \sum_f [I_{fijk} \times R_{fij}/R_{ij}]$). Alternatively, if richness (R_{ij}) data allows, BII can be calculated directly at the functional type level.

In the southern African example, at least three experts were interviewed *independently* for each broad taxonomic group. The interview process generated approximately 300 estimates per expert, and took about five hours per expert. Experts were not shown or informed about the magnitude of the estimates provided by other experts; neither were they jointly interviewed with the aim of generating a consensus estimate. All expert estimates were treated as valid and entered into the database, in order to enable us to determine the uncertainty around the estimates. The range of estimates obtained in this way was used to derive a confidence interval for the I_{ijk} estimates.

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Appendix 2: The BII algorithm and partial forms

The Biodiversity Intactness Index (BII) gives the average change in population size of the species of all well-described taxa in a particular area:

$$\mathrm{BII} = \frac{\sum_i \sum_j \sum_k \mathrm{R_{ij} A_{jk} I_{ijk}}}{\sum_i \sum_j \sum_k \mathrm{R_{ij} A_{jk}}}$$

where

R_{ij} = Richness of taxon *i* in ecosystem *j*

A_{jk} = Area of land use *k* in ecosystem *j*

I_{ijk} = Population size of taxon *i* under use *k* in ecosystem *j* / population size prior to transformation

BII can be disaggregated to successive levels of detail along different axes. The average change in population size for a specific taxon *i*, a particular ecosystem *j* or a particular land use *k* are respectively given by:

$$\mathrm{BII_i} = \frac{\sum_j \sum_k \mathrm{R_{ij} A_{jk} I_{ijk}}}{\sum_j \sum_k \mathrm{R_{ij} A_{jk}}} \qquad \mathrm{BII_j} = \frac{\sum_i \sum_k \mathrm{R_{ij} A_{jk} I_{ijk}}}{\sum_i \sum_k \mathrm{R_{ij} A_{jk}}} \qquad \mathrm{BII_k} = \frac{\sum_i \sum_j \mathrm{R_{ij} A_{jk} I_{ijk}}}{\sum_i \sum_j \mathrm{R_{ij} A_{jk}}}$$

BII can further be calculated for particular combinations of taxa, ecosystems or land use:

$$\mathrm{BII_{ij}} = \frac{\sum_k \mathrm{A_{jk} I_{ijk}}}{\sum_k \mathrm{A_{jk}}} \qquad \mathrm{BII_{jk}} = \frac{\sum_i \mathrm{R_{ij} I_{ijk}}}{\sum_i \mathrm{R_{ij}}} \qquad \mathrm{BII_{ik}} = \frac{\sum_j \mathrm{R_{ij} A_{jk} I_{ijk}}}{\sum_j \mathrm{R_{ij} A_{jk}}}$$

Appendix 3: Expert-derived population impact estimates

The mean expert-derived population impact of different land-use activates for different taxa, listed per biome, together with standard errors, used in the calculation of the BII.

		PROTECTED		MODERATE USE		DEGRADED		CULTIVATED		URBAN		PLANTATION	
		mean	*st err*	*mean*	*st err*	*mean*	*st err*	*mean*	*st err*	*mean*	*st err*	*mean*	*st err*
PLANTS	**ALL BIOMES**	**1.000**	**0.000**	**0.920**	**0.014**	**0.546**	**0.027**	**0.200**	**0.010**	**0.078**	**0.007**	**0.248**	**0.010**
	Forest	1.000	0.000	0.871	0.044	0.479	0.059	0.196	0.043	0.060	0.012		
	Fynbos	1.000	0.000	0.939	0.036	0.508	0.110	0.189	0.010	0.076	0.024	0.237	0.014
	Grassland	1.000	0.000	0.936	0.032	0.481	0.047	0.222	0.032	0.088	0.006	0.258	0.012
	Savanna	1.000	0.000	0.947	0.053	0.589	0.054	0.201	0.015	0.087	0.013		
	Shrubland	1.000	0.000	0.889	0.023	0.549	0.032	0.201	0.023	0.078	0.020		
	Wetland	1.000	0.000	0.940	0.022	0.669	0.056	0.193	0.028				
MAMMALS	**ALL BIOMES**	**0.998**	**0.001**	**0.791**	**0.028**	**0.472**	**0.042**	**0.361**	**0.045**	**0.207**	**0.031**	**0.453**	**0.090**
	Forest	0.994	0.006	0.823	0.018	0.580	0.051	0.398	0.041	0.238	0.015		
	Fynbos	1.000	0.000	0.932	0.024	0.696	0.161	0.491	0.157	0.196	0.083	0.512	0.163
	Grassland	0.998	0.002	0.691	0.088	0.372	0.028	0.211	0.025	0.150	0.072	0.395	0.102
	Savanna	0.997	0.003	0.756	0.044	0.405	0.064	0.449	0.032	0.201	0.055		
	Shrubland	1.000	0.000	0.719	0.064	0.447	0.078	0.384	0.170	0.249	0.124		
	Wetland	1.000	0.000	0.826	0.071	0.331	0.021	0.231	0.113				
BIRDS	**ALL BIOMES**	**0.996**	**0.002**	**0.998**	**0.017**	**0.571**	**0.048**	**0.736**	**0.060**	**0.568**	**0.038**	**0.515**	**0.087**
	Forest	0.991	0.009	0.987	0.008	0.711	0.139	0.650	0.090	0.477	0.044		
	Fynbos	1.000	0.000	0.949	0.032	0.648	0.155	0.898	0.062	0.576	0.118	0.639	0.116
	Grassland	0.991	0.009	1.004	0.044	0.553	0.029	0.779	0.026	0.575	0.046	0.391	0.097
	Savanna	0.995	0.005	0.998	0.027	0.542	0.122	0.708	0.115	0.527	0.133		
	Shrubland	1.000	0.000	1.085	0.067	0.542	0.158	1.045	0.105	0.683	0.057		
	Wetland	1.000	0.000	0.965	0.014	0.430	0.096	0.334	0.018				
REPTILES	**ALL BIOMES**	**0.994**	**0.003**	**0.946**	**0.010**	**0.720**	**0.023**	**0.374**	**0.017**	**0.232**	**0.017**	**0.314**	**0.023**
	Forest	0.989	0.011	0.951	0.015	0.754	0.047	0.414	0.044	0.197	0.025		
	Fynbos	0.991	0.003	0.957	0.027	0.704	0.069	0.362	0.031	0.292	0.049	0.281	0.028
	Grassland	1.000	0.000	0.921	0.039	0.736	0.033	0.384	0.039	0.207	0.008	0.347	0.030
	Savanna	1.000	0.000	0.961	0.029	0.722	0.053	0.382	0.046	0.210	0.028		
	Shrubland	0.985	0.015	0.950	0.022	0.699	0.077	0.384	0.072	0.252	0.055		
	Wetland	1.000	0.000	0.936	0.020	0.706	0.086	0.320	0.018				
AMPHIBIA	**ALL BIOMES**	**1.000**	**0.000**	**1.035**	**0.060**	**0.667**	**0.061**	**0.556**	**0.087**	**0.407**	**0.096**	**0.182**	**0.034**
	Forest	1.000	0.000	0.968	0.032	0.487	0.118	0.306	0.084	0.228	0.079		
	Fynbos	1.000	0.000	0.915	0.085	0.703	0.131	0.535	0.258	0.411	0.247	0.211	0.063
	Grassland	1.000	0.000	1.036	0.036	0.613	0.247	0.328	0.096	0.173	0.088	0.154	0.030
	Savanna	1.000	0.000	1.000	0.000	0.763	0.083	0.736	0.031	0.430	0.111		
	Shrubland	1.000	0.000	1.325	0.349	0.744	0.218	0.974	0.345	0.791	0.348		
	Wetland	1.000	0.000	0.967	0.033	0.689	0.141	0.459	0.133				

Appendix 4: Taxon functional groups and richness

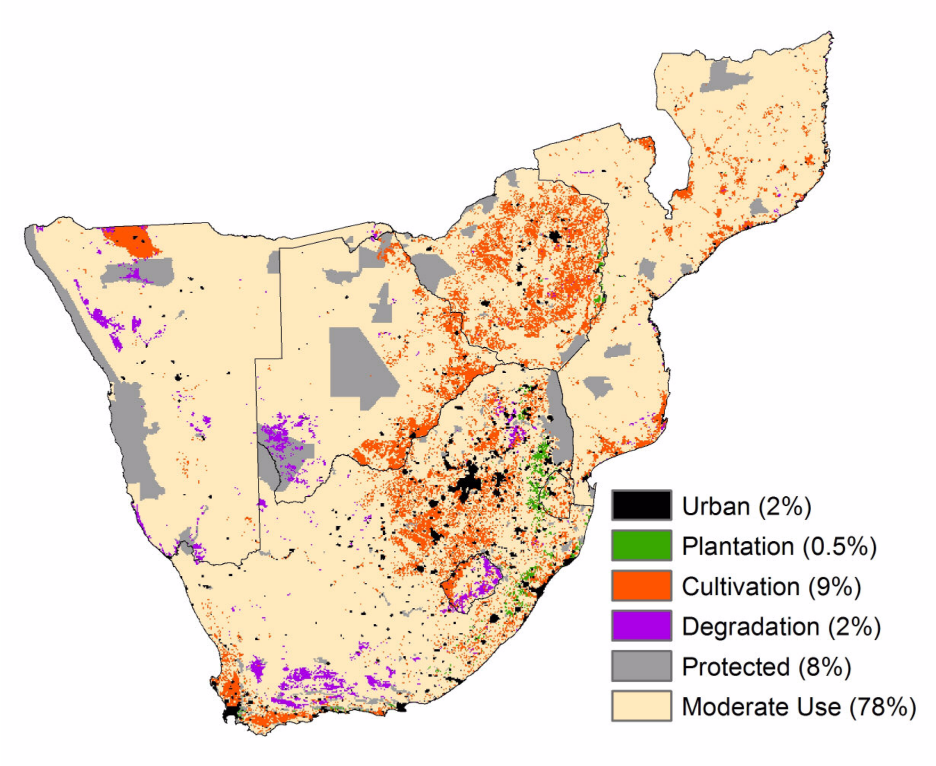
The functional groups defined for each taxon in order to collect the population impact data by means of expert interviews. Species richness per functional group per biome was used to weight the individual expert estimates up to the broad taxon level in calculating the BII. Note that the plant richness data are given as fractions of the total richness in each biome. Mike Rutherford is thanked for the plant richness estimates. James Harrison is thanked for the base data used to derive the richness of mammals, birds, reptiles and amphibia in each biome.

	SPECIES RICHNESS					
	Forest	**Savanna**	**Grassland**	**Shrubland**	**Fynbos**	**Wetland**
PLANTS						
Trees: valuable timber or fuel	0.35	0.10	0.01	0.00	0.01	0.01
Trees or shrubs: fruit or symbolic	0.10	0.10	0.00	0.00	0.01	0.01
Trees or shrubs:other	0.35	0.15	0.05	0.03	0.19	0.04
Graminoids: decreasers	0.01	0.14	0.30	0.14	0.05	0.12
Graminoids: increaser II	0.01	0.14	0.15	0.02	0.05	0.12
Graminoids: annuals/weedy	0.01	0.03	0.05	0.04	0.01	0.12
Geophytes	0.01	0.05	0.10	0.15	0.20	0.01
Forbs:weedy	0.01	0.05	0.01	0.05	0.03	0.12
Forbs: non-weedy	0.05	0.20	0.29	0.57	0.45	0.31
Bryophytes, ferns, epiphytes, lianas	0.10	0.03	0.05	0.00	0.02	0.12
TOTAL	**1**	**1**	**1**	**1**	**1**	**1**
MAMMALS						
Large herbivores (>5kg)	4	22	7	8	7	3
Small herbivores	10	12	9	15	8	0
Megaherbivores	1	5	0	0	0	1
Rodents	5	40	22	27	15	2
Bats	23	60	9	8	7	0
Large predators (>5 kg)	1	10	5	4	4	0
Small predators (<5kg)	6	23	12	14	12	3
Insectivores	13	9	6	4	0	0
Primates	4	5	1	1	1	0
TOTAL	**67**	**186**	**71**	**81**	**54**	**9**

	SPECIES RICHNESS					
	Forest	**Savanna**	**Grassland**	**Shrubland**	**Fynbos**	**Wetland**
BIRDS						
Large birds of prey	5	42	25	20	19	3
Small birds of prey	5	37	23	19	14	4
Gamebirds	2	32	23	18	12	2
Seedeaters	16	71	42	33	21	15
Waterfowl	4	30	28	25	24	32
Waders	4	69	57	43	41	58
Fruit eaters	28	40	14	15	11	0
Insect eaters	50	221	132	113	75	25
Sunbirds	8	13	5	6	5	0
TOTAL	**122**	**555**	**349**	**292**	**222**	**139**
REPTILES						
Large snakes (python)	0	1	0	0	0	1
Medium snakes	6	13	3	4	5	1
Small snakes	10	41	21	19	20	10
Burrowing snakes	4	26	7	7	3	0
Rock dwellers	2	29	18	20	12	0
Ground dwellers	10	49	28	50	30	0
Tree dwellers	10	14	2	6	5	0
Fossorial skinks, worm lizards	1	12	0	0	0	0
Monitors	0	2	2	1	0	1
Tortoises	1	8	2	7	5	0
Terrapins	0	5	1	1	1	5
Crocodiles	0	1	0	0	1	1
TOTAL	**44**	**201**	**84**	**115**	**82**	**19**
AMPHIBIA						
Water breeding, specialist, widespread	1	2	2	0	1	3
Water breeding, non specialist, widespread	16	43	21	13	15	52
Non-water breeding, specialist, widespread	8	10	5	3	6	9
Water breeding, specialist, restricted	9	0	1	0	6	10
Water breeding, non specialist, restricted	1	0	5	2	7	14
Non-water breeding, specialist, restricted	5	0	3	1	5	8
TOTAL	**40**	**55**	**37**	**19**	**40**	**96**

Appendix 5: Land use map for southern Africa

A land use map for southern Africa was compiled by combining data from several sources (see Table 3.1). Where land use categories from different data sources overlapped, land use was assigned in the following order of priority: urban, plantation, cultivated, degraded, protected. All areas not falling within one of these categories were assigned as moderate use. Numbers in brackets indicate the percentage of the region within each category.



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