Landscape Ecol (2012) 27:1121–1134
DOI 10.1007/s10980-012-9761-0

RESEARCH ARTICLE

Expanding the conservation toolbox: conservation planning of multifunctional landscapes

Belinda Reyers · Patrick J. O'Farrell · Jeanne L. Nel · Kerrie Wilson

Received: 24 November 2011 / Accepted: 15 May 2012 / Published online: 14 July 2012
© Springer Science+Business Media B.V. 2012

Abstract An area of convergence appears to be emerging between the approaches of conservation planning and the concepts of multifunctional landscapes, which if exploited correctly may assist in overcoming the resource and other constraints faced by biodiversity conservation, while at the same time furthering the aims of multifunctional landscapes to improve production abilities and overall sustainability. Using a multi-zone conservation planning approach, we explore the conservation costs, benefits to biodiversity conservation and possible ecosystem service payments associated with various land-use configurations, in the Little Karoo of South Africa, in order to develop and showcase a multifunctional landscape planning approach and its data requirements, as well as the possible cost savings to conservation agencies. The study uses four conservation planning scenarios, five land-use types, their conservation costs and biodiversity benefits, as well as possible payments from carbon sequestration and tourism. We find that the costs and biodiversity benefits associated with different land-uses varies substantially between land-uses, and also spatially within a land-use type. By incorporating this variation into a multi-zone conservation planning approach land-uses can be allocated in a way that achieves biodiversity targets while at the same time reducing costs by up to 50 % when compared with traditional binary approaches to conservation. Despite some challenges presented by cost and ecosystem service value data and the determination of land-use impacts on biodiversity and ecosystem services, the ability of conservation planning approaches to reflect differential contributions of particular land-uses to biodiversity targets and ecosystem services holds much potential for conservation planning, for multifunctional landscape objectives and for growing the resources and partnerships available to the establishment of sustainable and resilient landscapes.

Keywords Land-use · Conservation costs · Payments for ecosystem services · Carbon · Tourism · Grazing

Electronic supplementary material The online version of this article (doi:10.1007/s10980-012-9761-0) contains supplementary material, which is available to authorized users.

B. Reyers (✉) · P. J. O'Farrell · J. L. Nel
Natural Resources and the Environment, Council for Scientific and Industrial Research, PO Box 320, Stellenbosch 7599, South Africa
e-mail: breyers@csir.co.za

B. Reyers
Department of Conservation Ecology and Entomology, University of Stellenbosch, Private Bag X1, Stellenbosch 7602, South Africa

K. Wilson
School of Biological Sciences, University of Queensland, Brisbane St Lucia, QLD 4072, Australia

Introduction

The scale of the human enterprise and resultant biodiversity losses drive much of the global effort to conserve biodiversity. However, efforts to conserve biodiversity are constrained by competing and often conflicting agendas for social and economic development (Sanderson and Redford 2003; Reyers et al. 2010), low levels of political support, complex governance and institutional arrangements (Lombard et al. 2010), and existing and future budget shortfalls (James et al. 1999; Balmford et al. 2003; Bruner et al. 2004). In responding to these challenges the conservation community has expanded the approaches and tools at their disposal through for example developing more resource- and cost-efficient ways to conserve biodiversity e.g. off-reserve conservation (Binning 1997; Myers et al. 2000; Mace and Lande 1991; Wilson et al. 2009). Off-reserve conservation efforts focus on the management of whole landscapes, complementing formal protected areas with other (potentially cheaper) property rights mechanisms including the creation of management agreements and the designation of conservation easements (Hale and Lamb 1997; Pence et al. 2003). In addition to managing the demand on conservation budgets, more recently there has been an emergence of approaches that attempt to grow the available budget for conservation. Chief amongst these approaches are payment for ecosystem service (PES) schemes which are being supported by many conservationists because of their potential to conserve biodiversity, while unlocking additional funds through the protection and enhancement of ecosystem services (Reyers et al. 2012).

The array of ways to reduce demands on and grow conservation budgets has received much scientific attention and shows some promise in helping achieve conservation targets with limited resources (e.g. Pence et al. 2003; Naidoo and Ricketts 2006; Liu et al. 2008; Turpie et al. 2008). This new suite of off-reserve and ecosystem service aligned conservation tools has resulted in some major shifts in the way conservation is done. These shifts include a movement away from a focus on binary landscapes, where a parcel of land is either conserved entirely or not at all, to a more integrated view of biodiversity conservation as part of productive working landscapes where conservationists work in partnership with other sectors (agriculture, water and housing) to address not only biodiversity conservation objectives, but broader sustainability needs too. These more integrated approaches closely resemble the ideas put forward by landscape ecologists in the study and design of multifunctional landscapes. Multifunctional landscapes are landscapes which provide multiple environmental, social and economic functions and are able to achieve multiple societal needs including energy and food production, management of waste, conservation of biodiversity and the management of water quantity and quality across the landscape; the improvement of landscape heterogeneity and therefore resilience; and the provision of recreational opportunities (Lovell and Johnston 2009). While theoretically compelling, the field of multifunctional landscapes remains constrained by a lack of planning approaches able to: plan and establish multifunctional landscapes (Nassauer and Opdam 2008); consider the inherent costs and contributions of various parts of the landscape to multiple social, economic and ecological objectives (Lovell and Johnston 2009); and to account for synergies, threats and trade-offs between multiple functions in the landscape and objectives (Bennett et al. 2009).

New tools in the area of conservation planning (the systematic and spatially explicit identification of areas important to achieving particular biodiversity conservation targets (Margules and Pressey 2000)) including multiple land-use zonation (Wilson et al. 2010), ecosystem service data and planning approaches (Nelson et al. 2009; Reyers et al. 2009; Egoh et al. 2010) and approaches for including cost data into conservation planning (Carwardine et al. 2008) show much promise in addressing the above challenges identified in the study of multifunctional landscapes. There appears therefore to be an area of useful convergence between conservation planning and multifunctional landscapes, which if exploited correctly may assist not only overcoming the resource and other constraints faced by biodiversity conservation and its approaches, but may also further the aims of multifunctional landscapes to improve production abilities of land as well as their ecological functions and therefore the longer term resilience or sustainability of the landscape (Nassauer and Opdam 2008).

This study aims to take advantage of the convergence between multifunctional landscape thinking and conservation planning, as well as a long term biodiversity and ecosystem services research program in the Little Karoo of South Africa (Gallo et al. 2009;

Reyers et al. 2009; Egoh et al. 2010; O'Farrell et al. 2010) to explore the application of a recently developed conservation planning tool, together with data on land-use, biodiversity and ecosystem services, in order to develop and showcase a multifunctional landscape planning approach and its data requirements. Furthermore, it aims to evaluate the possible cost savings to conservation agencies through such engagements in a broader multifunctional landscape approach to conservation.

Methods

Study region

The Little Karoo region ($\sim$19 000 km^2) lies within a semi-arid intermontane basin and is the meeting point for three globally-recognized biodiversity hotspots (Fig. 1; Reyers et al. 2009; Egoh et al. 2010). The region, which is mostly comprised of privately owned farms, has a long history of extensive grazing and browsing by livestock, chiefly ostriches, but also sheep and goats. Very little (<10 %) clearing of natural habitat to cultivated land has taken place, while studies have pointed to the overarching role that overgrazing (mostly through intensive ostrich farming) has played in the degradation of vegetation and soil, biodiversity and ecosystem services (Reyers et al. 2009).

Analysis framework—Marxan with zones

We employed a version of the decision-support tool Marxan that deals with multiple land-use zone planning—Marxan with Zones (Watts et al. 2009). Marxan with Zones is based on an area selection algorithm that aims to meet pre-specified biodiversity conservation targets for the least cost, by allocating planning units (the spatial units of analysis—1 km^2 grid cells (n = 17,131)), and their biodiversity, to five different land-use zones. The land-use zones are the critical element around which this study is based in that they differ in their contribution to biodiversity conservation targets, the conservation actions required and associated costs to conservation agencies, and their potential for payments and other forms of income from ecosystem services. We defined five land-use zones for use in the Marxan analysis: "formal conservation", "private reserve", "conservancy", "grazing lands", and "cleared". Formal conservation areas were defined as state owned protected areas managed for biodiversity objectives and fall within the traditional on-reserve approach to conservation. The "private reserve", "conservancy" and "grazing lands" land-use zones corresponded with three different types of off-reserve areas with short term (20 year) management agreements between the land owner and the conservation agency concerning the use and management of the land. These agreements, in addition to management guidelines, allow for increasing levels of grazing of domestic and wild stock ranging from the sustainable levels recommended by the regional Department of Agriculture for "private reserves", twice the recommended sustainable levels in "conservancies" and four times the recommended rates (through use of feedlots, supplementary fodder and livestock camps) in "grazing lands" (Dean and MacDonald 1994). "Cleared" zones represent land cleared of natural vegetation for cultivation and urban development and do not contribute to biodiversity conservation objectives.

The analysis was divided into 3 phases (Fig. 2) beginning with an assessment of the initial state of each planning unit, which is used to determine the possible land-use zones into which that planning unit may be allocated and the actions required to put it into a particular land-use zone. This was followed by an analysis of the biodiversity contribution of each planning unit when allocated into each of the land-use zones, as well as the costs of this allocation. The final step applied the Marxan with Zones algorithm in identifying the best allocation of planning units per land-use zone in a configuration which best achieves the biodiversity conservation targets at minimum cost. Below we provide details on each of these steps, as well as their input datasets (Fig. 2).

Determining the initial state of the planning units

Marxan with Zones requires information on the initial state of each planning unit in order to determine the possible land-use zones to which it can be allocated. The initial state is also important as it determines the conservation actions required to allocate a planning unit to a particular land-use zone and the costs of such actions. To estimate initial state for each planning unit we used data on land cover to classify planning units as "protected pristine", "unprotected pristine",

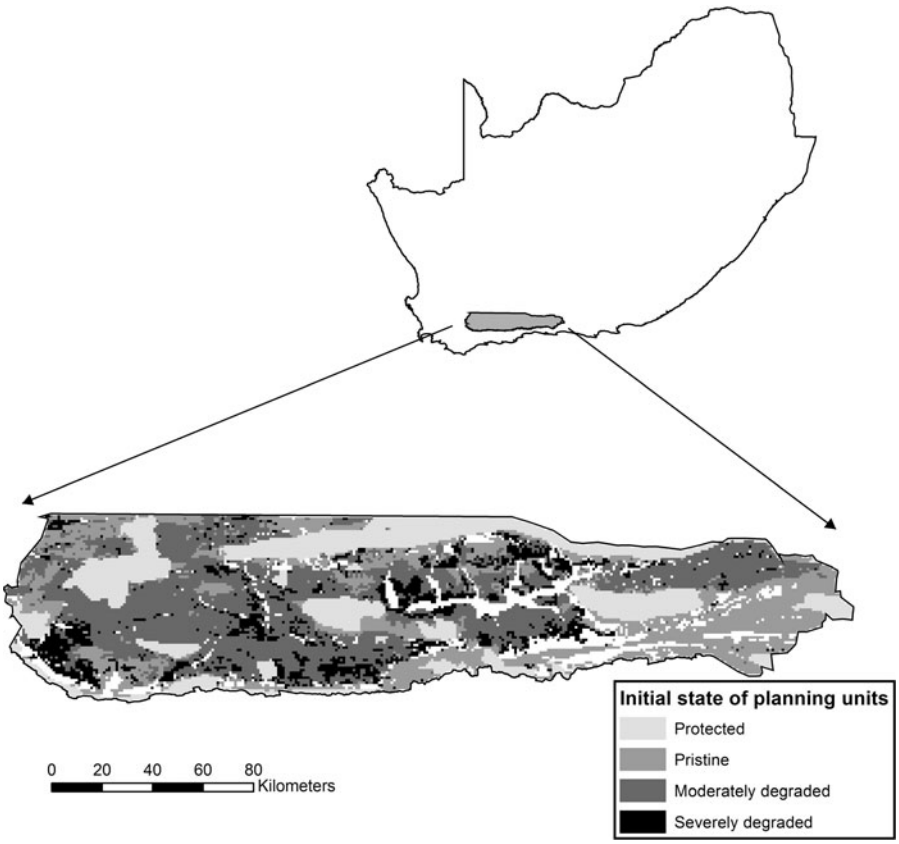

Fig. 1 The location of the Little Karoo study area in South Africa also illustrating the initial state of planning units. *White areas* are cleared by urban development and cultivation and are not included in the analysis

"moderately degraded" and "severely degraded" (Fig. 1). Land cover (the physical material on the land e.g. bare ground, trees) is used here to determine the state or condition of the land, it differs from land-use (i.e. how the land is utilised for socio-economic activities e.g. urban or agricultural uses) which is the focus of the Marxan with Zones analysis. Protected area coverage was used to differentiate pristine planning units already conserved. We then established rules for possible land-use zone allocations for each planning unit based on its initial state, together with the costs of each allocation (see supplementary material for details on classification and rules for planning unit allocations).

Biodiversity contributions and costs of land-use allocations

In order to measure the biodiversity contribution of each planning unit, Marxan with Zones requires data on the amount of biodiversity features (e.g. species, ecosystems) contained in each planning unit, as well as a quantitative target for each biodiversity feature. As a measure of biodiversity we used vegetation data from Vlok et al. (2005)—a fine-resolution, hierarchical map (1:50,000) which depicts 369 vegetation types grouped into biomes. Targets for the vegetation types were extracted from Gallo et al. (2009) and applied to each vegetation type. They were based on the quantitative species-turnover approach developed by Desmet and Cowling (2004) and range from 16 to 34 % of original extent of each vegetation type.

The five land-use zones differ in the condition or intactness of their biodiversity pattern and process and therefore how much a planning unit in a zone can contribute to the biodiversity targets. The "zone contribution value" employed in Marxan with Zones is a measure of this contribution to targets and varies between zero and 1, where a value of 1 implies that all biodiversity in that planning unit contributes to the target. We used biome level measures of biodiversity intactness extracted from Biggs et al. (2006) in determining the zone contribution values Table 1 (see supplementary material for details).

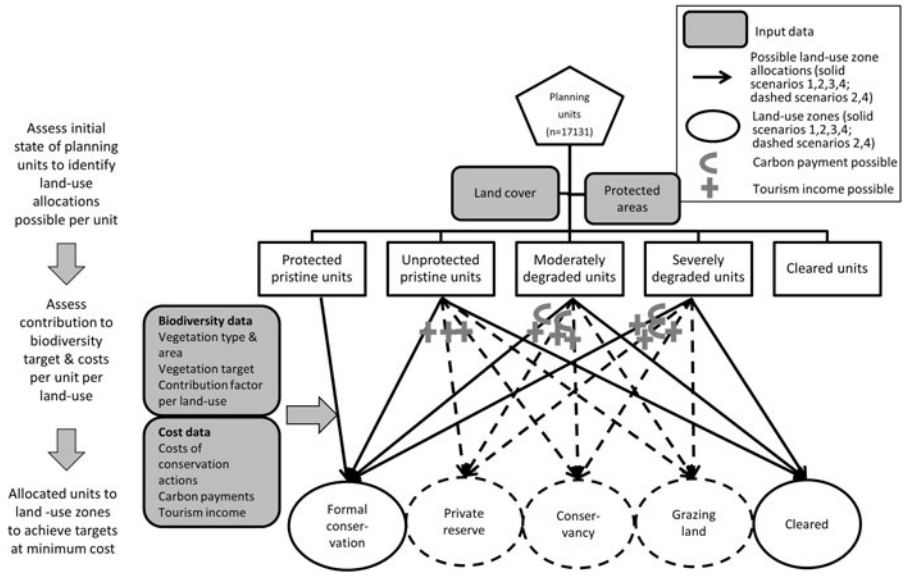

Fig. 2 Analytical framework used in the multiple land-use zone conservation planning process. Three phases of analysis are depicted starting with the determination of the initial state of the planning units used to determine possible land-use per planning unit. The second phase is the analysis of the planning unit specific biodiversity and cost data to determine the contribution and cost each planning unit makes when allocated to a land-use zone. The final phase allocated each planning unit to a particular land-use zone to meet biodiversity targets at least cost. The final phase compares four scenarios: Scenario 1 (*solid arrows*) uses only "formal conservation" and "cleared" zone, Scenario 2 (*solid and dashed arrows*) uses "formal conservation", all off-reserve zones and the "cleared" zone, Scenario 3 (*solid arrows*), pairs Scenario 1 with payments for ecosystem services through carbon sequestration and income generated through tourism to areas in "formal conservation" zones. Scenario 4 (*solid and dashed arrows*), pairs Scenario 2 with PES through carbon sequestration for areas moving into "formal conservation" and "private reserve" zones, and with income generated through tourism for all land-use allocations, except to "grazing lands". The symbols C and + illustrate planning unit allocations which can include carbon and tourism payments, respectively

Each allocation of a planning unit to a land-use zone involves a set of conservation actions (e.g. management, restoration) with associated costs from the perspective of a conservation agency. Costs could either be once off start-up costs (e.g. acquisition) or ongoing annual costs (e.g. management). The latter we endowed for 20 years at a discount rate of 3 %. We inflated all costs to 2008 values and converted all currencies into US$ values using a rate of 7.5 ZAR:1 US$. For some of the scenarios described below we also included opportunity costs of foregone grazing or cultivation. Table 2 contains the conservation actions and their costs used for each land-use allocation (details of the cost data are provided in the supplementary material).

In some of the scenarios below we included payments for carbon sequestration and tourism as a mechanism to reduce opportunity costs for land owners and the conservation agency. Data on these ecosystem services were based on 32 functional groupings of the vegetation types to determine carbon storage values and livestock numbers. Carbon storage data were based on a combination of field data and expert extrapolation extracted from Reyers et al. (2009). We calculated the payment for carbon sequestration based on the changes in the amount of carbon stored per planning unit for allocations of moderately and severely degraded planning units to "formal conservation" and "private reserve". The difference was based on the planning unit initial state and values of current carbon stored and the potential carbon stored if all areas were restored to pristine states at a rate of 4.2 t C/ha/year (Table 1; Mills and Cowling 2006). The high grazing levels in the "conservancy" and "grazing land" zones precluded these land-use zone allocation from carbon payments (Fig. 2). We used a value of US$30 per ton sequestered over the 20 year period or until the potential carbon levels are reached ((whichever is sooner) electronic supplementary material, Table S1).

To calculate the tourist income per planning unit we used data extracted from O'Farrell et al. (2010) on

Table 1 Contributions of each land-use zone to biodiversity targets and carbon storage functions

Biomes	Formal conserved	Private reserve	Conservancy	Grazing land
Contribution to biodiversity targets				
Fynbos	1.00	0.94	0.72	0.51
Thicket	1.00	0.95	0.77	0.59
Succulent Karoo	1.00	0.89	0.72	0.55
Wetlands	1.00	0.94	0.80	0.67
Carbon stored (t/C/ha)				
Thicket	195.00	146.67	103.33	N/A
Thicket mosaics	50.00	40.00	30.00	N/A
Succulent Karoo	25.00	12.50	5.00	N/A

Biodiversity contributions are adapted from Biggs et al. (2006), while carbon storage values are taken from Reyers et al. (2009)

Formal conserved areas are defined as state owned protected areas managed for biodiversity objectives. Private reserves are off-reserve areas with short term (20 year) management agreements which allow for grazing of domestic and wild stock at sustainable levels recommended by the regional Department of Agriculture. Conservancies are off-reserve areas with short term (20 year) management agreements which allow for grazing of domestic and wild stock at twice the recommended sustainable levels. Grazing lands are off-reserve areas with short term (20 year) management agreements which allow for grazing of domestic and wild stock at four times the recommended rates (through use of feedlots, supplementary fodder and livestock camps). Cleared zones represent land cleared of natural vegetation for cultivation and urban development and do not contribute to biodiversity conservation objectives

scenic tourism values (Table S1). We calculated this income for planning units within view of major tourist routes determined from a 10 km viewshed mapped by Reyers et al. (2009). This income was used to reduce the costs of planning unit allocations from pristine unprotected, moderately and severely degraded areas to "formal conservation", "private reserve" and "conservancy" (Fig. 2).

Scenarios

In order to assess the range and possible contribution of the set of conservation tools available we used Marxan with Zones to compare four conservation planning scenarios (Fig. 2). Scenario 1 was formulated using only two land-use zones ("formal conservation" and "cleared" zones) and corresponds with the conventional on-reserve binary approach to conservation planning (a planning unit either contributes fully to conservation targets, or not at all). Scenario 2 moved beyond binary to a multifunctional approach which includes "formal conservation", all off-reserve zones and the "cleared" zone, their contributions to targets (Table 1) and their varying costs (Table 2). Scenario 3 paired Scenario 1 with payments for ecosystem services (PES). It used only the binary options of "formal conservation" and "cleared" zones, but included the option for PES through carbon sequestration, as well as income generated through tourism to areas in "formal conservation" zones. Scenario 4, reflecting a combined approach pairing Scenario 2, the multifunctional approach, with PES, used all zones, and included the option for PES through carbon sequestration for areas moving into "formal conservation" and "private reserve" zones, allocations which together with allocations to "conservancy" zones could also generate income from tourism.

Results

Conservation contributions and costs across a landscape

Data adapted from Biggs et al. (2006) on the contribution of different land-use zones to biodiversity targets (Table 1) indicate that while formal conservation zones contribute 100 % to biodiversity targets, private reserves contribute on average 93 % to targets, followed by conservancies (75 %) and grazing land (58 %). Planning units in the Fynbos biome appear most sensitive to high levels of grazing with contributions ranging from 72 % (conservancies) to 51 % (grazing lands), while Succulent Karoo areas are most sensitive to low levels of grazing (89 % contribution to targets for private reserves).

In examining the distribution of areas suitable for carbon sequestration and tourism activities, the central lowlands of spekboom (*Portulacaria afra*) dominated

Table 2 Possible land-use zone allocations for planning units based on their initial state (determined by land cover and protected area data). The set of conservation actions and costs required for each allocation is shown

Initial state of planning unit	Future land-use zone				
	Formal conservation	Private reserve	Conservancy	Grazing land	Cleared
Protected pristine	OnM	Not possible	Not possible	Not possible	No cost
Unprotected pristine	Aq + OnM	OfM + MaxOp − InGraz (20 year)	OfM + MaxOp − InGraz2 (20 year)	OfM + MaxOp − InGraz4 (20 year)	No cost
Moderately degraded land	Aq + OnM	OfM + MaxOp − InGraz (15 year)	OfM + MaxOp − InGraz2 (15 year)	OfM + MaxOp − InGraz4 (15 year)	No cost
Severely degraded land	Aq + OnM + Rest	OfM + MaxOp − InGraz (15 year) + Rest	OfM + MaxOp − InGraz2 (15 year) + Rest	OfM + MaxOp − InGraz4 (15 year) + Rest	No cost

Cost equations for each possible land-use allocation (cost abbreviations: *OnM* on-reserve management, *OfM* off-reserve management, *Aq* acquisition, *Rest* restoration, *MaxOp* maximum opportunity cost, *InGraz* income from sustainable grazing at recommended levels, *InGraz2* income from intensive grazing at two times recommended levels, *InGraz4* income from intensive grazing at four times recommended levels)

thicket habitats (Fig. 3) were found to have high carbon sequestration potential. For these areas, placing a planning unit into the formal conservation land-use zone will sequester the most carbon due to the low levels of grazing. Allocating that planning unit to private reserves will result in decrease of 24 % in average carbon sequestration, while allocating the unit to conservancies results in 45 % declines. Some sites in the Succulent Karoo biome show some sequestration potential (Fig. 3) but changes in grazing intensity caused by land-use changes from formal conservation areas to private reserves or conservancies cause 50 and 79 % declines in this storage ability respectively. The tourism viewshed (Fig. 3) due to its method of determination lies within close proximity to tourism routes, especially those within scenic mountain areas. Planning units within the tourism viewshed could bring in income of $45,000 per planning unit. The high numbers of planning units outside of the viewshed resulted in a low average value of $12,000 which is much lower than most of the other conservation costs (Fig. 4). The income values for carbon had a high average value of $40,000 per planning unit and compared favourably even with high costs of acquisition ($\bar{x}$ = $65,000) and opportunity costs (= $35,000).

Land acquisition costs were the highest average conservation cost in the Little Karoo ($US65,000; Fig. 4) closely followed by the average for: maximum opportunity costs ($35,000), intensive grazing opportunity costs ($33,000), and on reserve management costs ($28,000). Cultivation opportunity costs were the lowest average cost at $6,000. However, acquisition and opportunity costs varied spatially (Figs. 3, 4) with acquisition costs ranging from $6,000 to $363,000 per planning unit and maximum opportunity costs ranging from $15,000 to $245,000 per planning unit. In fact, while the cultivation opportunity cost displayed a low average cost due to the preponderance of planning units with no cultivation potential, where cultivation is possible these planning units showed some of the highest costs ($245,000) and determined the maximum opportunity cost in these units. Maximum values for intensive grazing were lower at $60,000 per planning unit. The spatial variation in acquisition costs (determined at the district municipality level) showed low values for districts in the north, while districts in the south and west were the most expensive. Opportunity costs associated with intensive grazing and cultivation (Fig. 3b, c) show low cost mountainous areas and expensive lowland patches, especially close to rivers and productive land.

Restoration and management costs were fixed costs per planning unit and did not vary spatially. Restoration costs were $141,000 per planning unit across the region for planning units which were severely degraded, while on- and off-reserve management costs were $23,000 and $29,000 respectively. The cost of acquiring all planning units in the study area would total $1.1 billion, for compensating land owners for foregone intensive grazing opportunities $560 million, and for compensating foregone cultivation

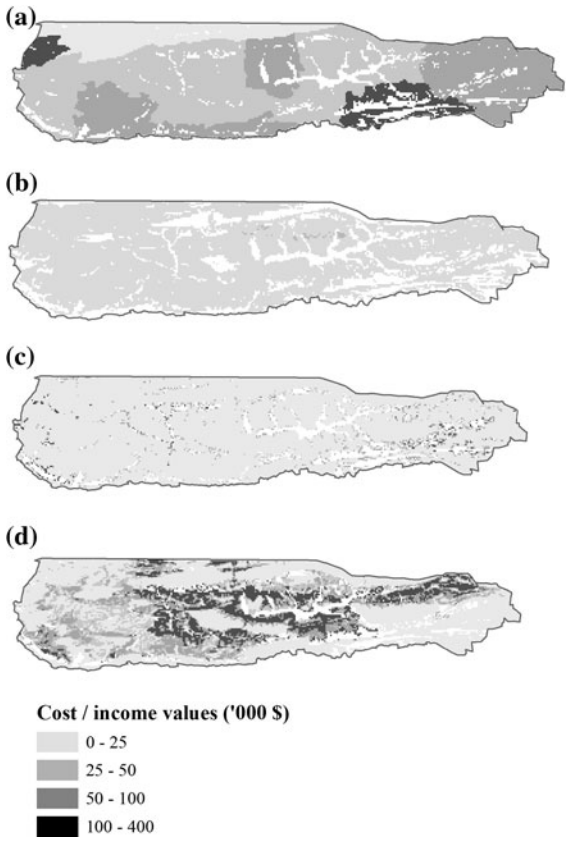

Fig. 3 Distribution of spatially variable cost and income values across the landscape. Values reflect the costs per planning unit for **a** acquisition, **b** compensating the land owner for foregone intensive grazing opportunities, **c** compensating the land owner for foregone cultivation opportunities, and **d** income values from carbon payments. Values are in '000 US$

opportunities $102 million. It would cost conservation agencies $495- and $395- million to manage all planning units on and off reserve respectively, and $289 million to restore all degraded planning units.

Scenario outputs

All scenarios were able to achieve targets set for vegetation types by at least 99 %. When comparing the outputs of Scenario 1 and 2, where the former only includes formal conservation areas and the latter includes all zones, findings show that compared to 43 % of the planning units selected by Scenario 1 for formal conservation, Scenario 2 allocated only 27 % of planning units into the formal conservation zone with 5, 6 and 11 % of planning units selected into "private reserves", "conservancy", and "grazing land" zones respectively. Scenario 2 selected more planning units; however the variety of zones available in Scenario 2 resulted in a solution that would cost the conservation agency $475 million, 12 % less than Scenario 1 ($542 million; Figs. 5, 6).

Scenario 3 resulted in 47 % of planning units allocated to the formal conservation zone, more planning units than the number selected in Scenario 1, but less than Scenario 2 (Fig. 5). With the inclusion of the option of payments from carbon and tourism in this scenario the total cost was $318 million, 41 % less than Scenario 1 and 33 % less than Scenario 2 (Fig. 6). Finally, Scenario 4 selected 50 % of the planning units, with more (29 and 13.5 %) planning units in the formal conservation zone and private reserves respectively than Scenario 2, and less (3.5 and 3.6 %) in conservancies and grazing lands respectively. The option for payments for carbon and tourism makes this the cheapest scenario at $ 229 million, 57.8 % less than Scenario 1 (Fig. 6).

Discussion

As this study and recent work by others (e.g. Wilson et al. 2007) demonstrates, conservation can be done in a multitude of ways with a multitude of costs and benefits which differ not only from one another, but also vary across space. While we only consider a small set of conservation actions and costs in the Little Karoo (acquisition, management, restoration and compensation) the variation of these in terms of their contribution to biodiversity conservation and their costs to conservation agencies is substantial, ranging by orders of magnitude within and between costs. For example allocating a planning unit to a grazing land-use zone in the little Karoo will protect 42 % less of its biodiversity than if the planning unit were placed in a formal conservation zone. Similarly buying a planning unit would cost an average of $65,000 plus management costs ($\bar{x}$ = $28,000), while managing it through an off-reserve agreement with the landowner would reduce costs to compensation costs of an average $33,000 and would also reduce the management costs by 20 %. By adding new approaches to conservation e.g. PES and tourism mechanisms even more variation in costs is encountered. Where possible, carbon payments of $40,000 per planning unit and tourism values of $45,000 per planning unit can reduce conservation costs significantly.

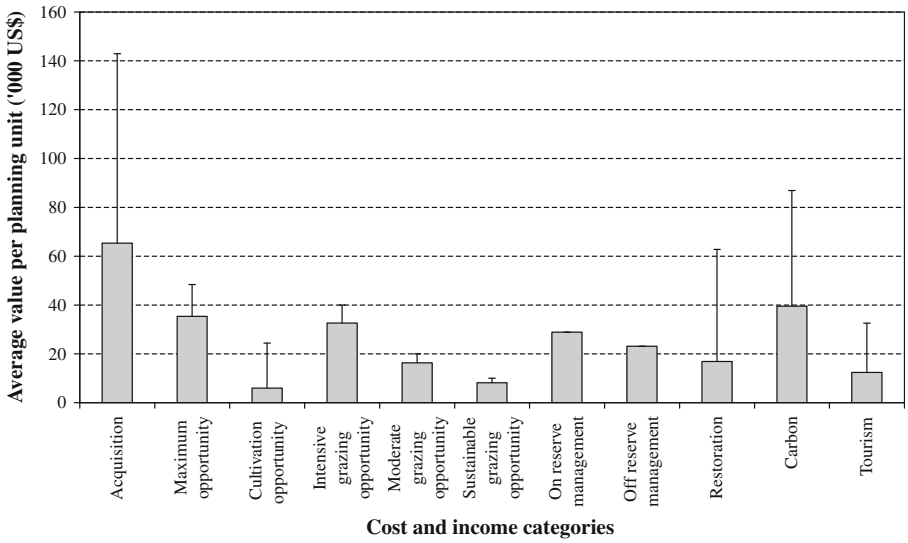

Fig. 4 Average (± SD) values in '000 US$ for conservation costs and income per planning unit

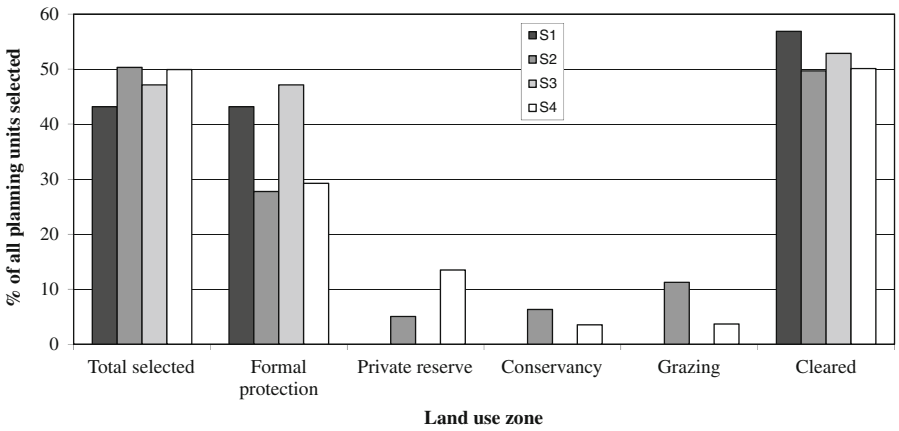

Fig. 5 Number of planning units selected per scenario per land-use zone and in total. Scenario 1 uses only "formal conservation" and "cleared" zone, Scenario 2 uses "formal conservation", all off-reserve zones and the "cleared" zone, Scenario 3, pairs Scenario 1 with payments for ecosystem services through carbon sequestration and income generated through tourism to areas in "formal conservation" zones. Scenario 4 pairs Scenario 2 with PES through carbon sequestration for areas moving into "formal conservation" and "private reserve" zones, allocations which together with allocations to "conservancy" zones, also receive payments from tourism

In comparing the different types of costs, we found that unlike other studies (e.g. Naidoo et al. 2006) there were large differences between acquisition ($\bar{x}$ = \$65,000) and opportunity costs ($\bar{x}$=\$35,000). This difference is most likely due to differences in resolution of the data used to calculate these costs, but also a result of important land-use drivers in the Little Karoo. The region has witnessed a recent move to "lifestyle farming" where farms are bought by urban residents as lifestyle investments or hobby farms rather than for production activities. This phenomenon has decoupled the acquisition cost of the farm from the land's production potential and has resulted in some positive impacts for conservation in the area through the recovery of many overgrazed and ploughed areas, but it has also had some negative consequences through the introduction of extra-limital species and through the intensification of production activities on other farms where land values are being driven up (O'Farrell et al. 2008). This study shows that shifts in land-use and the

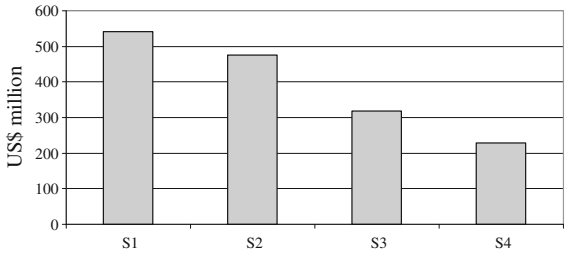

Fig. 6 Total cost of each scenario's selected set of planning units. Scenario 1 uses only "formal conservation" and "cleared" zone, Scenario 2 uses "formal conservation", all off-reserve zones and the "cleared" zone, Scenario 3, pairs Scenario 1 with payments for ecosystem services through carbon sequestration and income generated through tourism to areas in "formal conservation" zones. Scenario 4, pairs Scenario 2 with PES through carbon sequestration for areas moving into "formal conservation" and "private reserve" zones, allocations which together with allocations to "conservancy" zones could also generate income from tourism

decoupling of land value from production potential have substantial implications for conservation in the area, making traditional approaches to conservation very expensive and almost impossible under current conservation budgets (Scenario 1: $ 542 million).

By combining the variation in costs and benefits with a software platform, like Marxan with Zones, which is able to take the spatial variation of costs and benefits into account, as well as multiple land-use zones and multiple targets, the size of the cost reductions to conservation become apparent. Scenario 3, which involves the expensive acquisition of almost 50 % of the Little Karoo, costs a third of the price of Scenario 1 (which does not include payments from ecosystem services) if conservation budgets can be supplemented with payments for carbon sequestration and tourism income. These gains are even more noticeable when one moves beyond acquisition to off-reserve management approaches combined with income streams from carbon sequestration and tourism to offset the high opportunity costs in the area (Scenario 4). Scenario 4 demonstrates how one could meet all biodiversity targets and conserve nearly half of the Little Karoo at less than half the price it would cost to buy up the land in Scenario 1.

Conservation planning for multifunctional landscapes

From this study it appears that conservation planning tools like Marxan with Zones have the potential to help move conservation planning forward into a more integrated way of planning landscapes for conservation functions. But perhaps more importantly they also show potential in helping conservation and its approaches, engage more broadly with other stakeholders in the landscape, in planning landscapes with multiple functions, while at the same time contributing to the need for approaches for multifunctional landscape planning alluded to by Nassauer and Opdam (2008). These new conservation planning approaches can integrate knowledge of how landscapes are structured and function to recognise not only what land-use is best located where, but also what mutually inclusive multiple needs can be met by particular pieces of land and where trade-offs are unavoidable.

Ecologists and conservation biologists, due to their understanding of the biophysical structures and functions of landscapes, are key role players in the planning of multifunctional landscapes in order to ensure that they satisfy human demands for resources, while not undermining the ecological processes and other flows necessary for long term sustainability (Lovell and Johnston 2009). This need for ecological knowledge not only makes it possible for conservation biologists and ecologists to engage with other sectors and disciplines in planning multifunctional landscapes (e.g. forestry, Wilson et al. 2010), but also makes their data and tools useful to other sectors involved in the planning of multifunctional landscapes (e.g. land-use planning and water management; Theobald et al. 2000; Reyers et al. 2010).

While adopting a narrow conservation focus, this study has demonstrated the potential of this area of convergence between conservation planning and multifunctional landscapes and the possibility of configuring the landscape into multiple land-uses with differing costs and benefits to specific conservation objectives. However, if these tools are to reach their potential in the arena of conservation planning, but also in the more integrated planning of multifunctional landscapes, there are a few hurdles we encountered that need to be overcome—hurdles mostly to do with data requirements, but also with the complex context of implementation. We discuss some of these hurdles and possible solutions below based on our experiences in this study.

The cost and benefits of multifunctional landscapes

Conservation planning for multifunctional landscapes is data intensive, requiring data on the costs and

contributions of various land-uses to multiple targets (Lovell and Johnston 2009). As demonstrated in this study this requires not only data on the biodiversity features contained within a planning unit and the acquisition and management cost of the planning unit, it also requires information on land-use options, values and impacts, as well as trade-offs and synergies between biodiversity, ecosystem services and other land-use opportunities. Recent progress in ecology has already begun to contribute to this need providing data on the distribution of biodiversity, the distribution of ecological process and other flows and the location of areas supplying ecosystem services. Much of these data are useful to the conservation planning process. However, many of these datasets are collected or modelled at one point in time, and thus the determination of trends and trade-offs remains a challenge. Furthermore, many ecosystem service datasets reflect a potential biophysical measure (e.g. water run-off or timber biomass) rather than an actual measure of the use of ecosystem services, limiting the determination of benefits or values of such services.

But perhaps the biggest hurdle we encountered was determining the contributions of different land-uses to biodiversity targets and ecosystem service values. Even in this data rich study area, the analysis was constrained to conservation and grazing associated land-uses due to a lack of information on the impacts of other possible land-uses, e.g. croplands, on biodiversity and ecosystem services. The limited data on grazing land-use impacts on ecosystem services meant the list of ecosystem services was quite constrained from the many datasets used in Reyers et al. (2009) and Egoh et al. (2010) to only carbon storage and tourism. In the case of the latter the service was rather crudely included as a binary layer.

There is a clear need for more work in the area of land-use impacts and trade-offs in biodiversity and ecosystem services, especially in the area of intensive land-uses like urban development, mining and agriculture (e.g. Gordon et al. 2010), land-uses which are increasing in extent and impact due to heightened global demands.

The suite of planning units and conservation actions selected has major implications for the cost and therefore feasibility of the ultimate conservation network identified. This requires detailed and accurate cost data. In this study challenges around resolution and scale of cost data, as well as the need to extrapolate from the limited studies generating these data, make these findings limited to demonstrating the range of costs and relative costs, rather than the actual costs. While conservation planners have recognised the value of cost data to conservation planning for some time now (e.g. Ando et al. 1998; Balmford et al. 2003) and developed useful approaches for including cost data into conservation planning (e.g. Carwardine et al. 2008), it is only recently that the complexity of conservation costs has really begun to be recognised by the conservation planning community (Naidoo et al. 2006; Carwardine et al. 2010).

There has been a concerted global effort to assemble and test a wide variety of biodiversity data sets and approaches for conservation planning, however less attention has been paid to gathering and assessing data on the costs of conservation and developing best practice for the incorporation of such data into conservation plans. As Moore et al. (2004) point out cost is at the heart of most complementarity based approaches to conservation, therefore the empirical weaknesses in conservation cost data are a cause for concern. Furthermore, as Carwardine et al. (2010) highlight, data on the costs of conservation action are also inherently uncertain and can vary with individual perceptions, values and markets. These multiple sources of uncertainty surrounding cost data in conservation planning supports calls to invest the same amount of research effort into collecting, assessing, testing surrogates and integrating economic data as we have over the past decades to biodiversity data (Naidoo et al. 2006). Furthermore, if these conservation planning approaches are to be useful in multifunctional landscape planning, then there is a need to go beyond conservation costs to include other costs of land-use management from agriculture, water, urban development and other sectors.

In this study we identified ecosystem service valuation and payments as a final data gap in the planning of multifunctional landscapes. While there has been some good progress estimating the benefits and values of a suite of ecosystem services from global to local scales (e.g. O'Farrell et al. 2011), in the case of this study we were limited to carbon sequestration payments and tourism incomes. Other services of importance in this area e.g. water flow regulation, soil protection, flood protection and pollination remain challenging to assign an economic value to either because of an absence of methods with which to do so

(e.g. water flow regulation) or because of an absence of markets which implies that the value remains unrealised (e.g. soil protection). Advances in methods for valuation (e.g. TEEB 2010) and in understanding markets (e.g. Turpie et al. 2008) hold some promise for moving this area of conservation planning for multifunctional landscapes forward, but it remains unlikely that all types of ecosystem services will be amenable to inclusion in these sorts of cost-benefit analyses.

Finally the outputs generated by platforms such as Marzan with Zones, are but one stage in the planning process and implementing the results remains an enormous challenge (Reyers et al. 2010). The challenges of implementing conservation plans have been much explored (Cowling et al. 2008), but by engaging beyond just conservation functions and objectives a multifunctional landscape approach to conservation planning could help overcome some of the implementation challenges, especially those associated with societal relevance, resource constraints and value systems (Reyers et al. 2010). For example in exploring possible revenue streams required for implementation, Payment for Ecosystem Service (PES) in the traditional sense of willing buyer, willing seller are proving challenging to implement. However when teamed with start up public funding, poverty relief programs and political will they have seen successes in South Africa in clearing invasive alien plants and restoring hydrological functions (Turpie et al. 2008) and in the design of internationally and nationally funded restoration programs (Blignaut et al. 2008).

The expanded conservation toolbox

This study has demonstrated and measured the possible contributions of a multifunctional landscape approach to conserving biodiversity and ecosystem services in the Little Karoo. Although focused on the conservation sector, through a set of realistic scenarios, it has made clear that there are substantial reductions to costs and growth in potential income streams possible through the application of a multifunctional landscape lens to encourage off-reserve mechanisms, pay landowners for foregone opportunities through alternative income streams and supplement conservation budgets with these income streams. Moving beyond the conservation sector and its limited budgets, the growing interest in the societal costs of inappropriate land-use management and its impacts on flood regimes, water shortages and changes in water quality, when included in the planning of multifunctional landscapes, opens up more avenues for exploring ways to grow the resources and partnerships available to sustainable and multifunctional landscape planning and implementation.

Acknowledgments We thank Willem de Lange for his insights into conservation costs. We also thank Richard Cowling and Jan Vlok for initiating this project. The Council for Scientific and Industrial Research, National Research Foundation of South Africa and Australian Research Council are thanked for their financial support.

References

Ando A, Camm J, Polasky S, Solow A (1998) Species distributions, land values, and efficient conservation. Science 279:2126–2128

Balmford A, Gaston KJ, Blyth S, James A, Kapos V (2003) Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. Proc Natl Acad Sci USA 100:1046–1050

Bennett EM, Peterson GD, Gordon LJ (2009) Understanding relationships among multiple ecosystem services. Ecol Lett 12:1394–1404

Biggs R, Reyers B, Scholes RJ (2006) A biodiversity intactness score for South Africa. S Afr J Sci 102:277–283

Binning C (1997) Beyond reserves: options for achieving nature conservation objectives in rural landscapes. In: Klomp N, Lunt I (eds) Frontiers in ecology: building the links. Elsevier, Oxford, pp 155–168

Blignaut J, Aronson J, Mander M, Marais C (2008) Investing in natural capital and economic development: South Africa's Drakensberg Mountains. Ecol Restor 26:143–150

Bruner AG, Gullison RE, Balmford A (2004) Financial costs and shortfalls of managing and expanding protected-areas systems in developing countries. Bioscience 54:1119–1126

Carwardine J, Wilson KA, Ceballos G, Ehrlich PR, Naidoo R, Iwamura T, Hajkowicz S, Possingham HP (2008) Cost-effective priorities for global mammal conservation. Proc Natl Acad Sci USA 105:11446–11450

Carwardine J, Wilson KA, Hajkowicz SA, Smith RJ, Klein CJ, Watts M, Possingham HP (2010) Conservation planning when costs are uncertain. Conserv Biol 24:1529–1537

Cowling RM, Egoh B, Knight AT, O'Farrell P, Reyers B, Rouget M, Roux D, Welz A, Wilhelm-Rechman A (2008) An operational model for mainstreaming ecosystem services for implementation. Proc Natl Acad Sci USA 105:9483–9488

Dean WRJ, Macdonald IAW (1994) Historical changes in stocking rates of domestic livestock as a measure of semi-arid and arid land degradation in the Cape Province, South Africa. J Arid Environ 26:282–298

Desmet PG, Cowling RM (2004) Using the species-area relationship to set baseline targets for conservation. Ecol Soc 9:11

Egoh BN, Reyers B, Carwardine J, Bode M, O'Farrell PJ, Wilson KA, Possingham HP, Polasky S, Rouget M, De Lange W, Cowling RM (2010) Safeguarding biodiversity and ecosystem services in South Africa's Little Karoo: trade-offs and synergies. Conserv Biol 24:1021–1030

Gallo JA, Pasquini L, Reyers B, Cowling RM (2009) The role of private conservation areas in biodiversity representation and target achievement within the Little Karoo region, South Africa. Biol Conserv 142:446–454

Gordon LJ, Finlayson CM, Falkenmark M (2010) Managing water in agriculture for food production and other ecosystem services. Agric Water Manag 97:512–519

Hale P, Lamb D (1997) Conservation outside nature reserves. Centre for Conservation Biology, The University of Queensland, Brisbane

Hoekstra JM, Boucher TM, Ricketts TH, Roberts C (2005) Confronting a biome crisis: global disparities of habitat loss and protection. Ecol Lett 8:23–29

James AN, Gaston KJ, Balmford A (1999) Balancing the Earth's accounts. Nature 401:323–324

Liu J, Li S, Ouyang Z, Tam C, Chen X (2008) Ecological and socioeconomic effects of China's policies for ecosystem services. Proc Natl Acad Sci USA 105:9477–9482

Lombard AT, Cowling RM, Vlok JHJ, Fabricius C (2010) Designing conservation corridors in production landscapes: assessment methods, implementation issues, and lessons learned. Ecol Soc 15:7. http://www.ecologyandsociety.org/vol15/iss3/art7/

Lovell ST, Johnston DM (2009) Creating multifunctional landscapes: how can the field of ecology inform the design of the landscape? Front Ecol Environ 7:212–220

Mace GM, Lande R (1991) Assessing extinction threats: toward a re-evaluation of IUCN threatened species categories. Conserv Biol 5:148–157

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405:243–253

Mills AJ, Cowling RM (2006) Rate of carbon sequestration at two thicket restoration sites in the Eastern Cape, South Africa. Restor Ecol 14:38–49

Moore J, Balmford A, Allnutt T, Burgess N (2004) Integrating costs into conservation planning across Africa. Biol Conserv 117:343–350

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Naidoo R, Ricketts TH (2006) Mapping the economic costs and benefits of conservation. PLoS Biol 4(11):e360. doi: 10.1371/journal.pbio.0040360

Naidoo R, Balmford A, Ferraro PJ, Polasky S, Ricketts TH, Rouget M (2006) Integrating economic costs into conservation planning. Trends Ecol Evol 21:681–687

Nassauer JI, Opdam P (2008) Design in science: extending the landscape ecology paradigm. Landscape Ecol 23:633–644

Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron DR, Chan KMA, Daily GC, Goldstein J, Kareiva P, Lonsdorf E, Naidoo R, Ricketts TH, Shaw MR (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ 7:4–11

O'Farrell PJ, Le Maitre DC, Gelderblom C, Bonora D, Hoffman T, Reyers B (2008) Applying a resilience framework in the pursuit of sustainable land-use development in the Little Karoo, South Africa. In: Burns ME, Weaver A (eds) Exploring sustainability science—a Southern African perspective. SUN Press, Stellenbosch, pp 383–430

O'Farrell PJ, Reyers B, Le Maitre DC, Milton SJ, Egoh B, Maherry A, Colvin C, Atkinson D, De Lange W, Blignaut JN, Cowling RMC (2010) Multi-functional landscapes in semi arid environments: implications for biodiversity and ecosystem services. Landscape Ecol 25:1231–1246

O'Farrell P, Le Maitre D, De Lange W, Reyers B, Blignaut J, Milton S, Atkinson D, Egoh B, Maherry A, Colvin C, Cowling R (2011) The possibilities and pitfalls presented by ecosystem service valuation in an arid biodiversity hotspot. J Arid Environ 75:612–623

Pence GQK, Botha MA, Turpie JK (2003) Evaluating combinations of on- and off-reserve conservation strategies for the Agulhas Plain, South Africa: a financial perspective. Biol Conserv 112:253–274

Reyers B, O'Farrell PJ, Cowling RM, Egoh BN, Le Maitre DC, Vlok JHJ (2009) Ecosystem services, land-cover change, and stakeholders: finding a sustainable foothold for a semiarid biodiversity hotspot. Ecol Soc 14(1):38. http://www.ecologyandsociety.org/vol14/iss1/art38/

Reyers B, Roux DJ, Cowling RM, Ginsburg AE, Nel JL, O'Farrell P (2010) Putting conservation plans to work: conservation planning as a transdisciplinary process. Conserv Biol 24:957–965

Reyers B, Polasky S, Tallis H, Mooney H, Larigauderie A (2012) Finding a common ground for biodiversity and ecosystem services. BioScience 62(5):503–507

Rodrigues ASL, Andelman SJ, Bakarr MI, Boitani L, Brooks TM, Cowling RM, Fishpool LDC, Fonseca GAB, Gaston KJ, Hoffmann M, Long JS, Marquet PA, Pilgrim JD, Pressey RL, Schipper J, Sechrest W, Stuart SN, Underhill LG, Waller RW, Watts MEJ, Yan X (2004) Effectiveness of the global protected area network in representing species diversity. Nature 428:640–643

Sanderson SE, Redford KH (2003) Contested relationships between biodiversity conservation and poverty alleviation. Oryx 37:389–390

TEEB (2010) Mainstreaming the economics of nature: a synthesis of the approach, conclusions and recommendations of TEEB. Progress Press, Malta

Theobald DM, Hobbs NT, Bearly T, Zack JA, Shenk T, Riebsame WE (2000) Incorporating biological information in local landuse decision making: designing a system for conservation planning. Landscape Ecol 15:35–45

Turpie JK, Marais C, Blignaut JN (2008) The working for water program: evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecol Econ 65:788–798

Vlok JHJ, Cowling RM, Wolf T (2005) A vegetation map for the Little Karoo. Unpublished maps and report for a SKEP project supported by Grant No 1064410304. Critical Ecosystem Partnership Fund, Cape Town. http://bgis.sanbi.org/littlekaroo/index.asp. Accessed Jan 2008

Watts ME, Ball IR, Stewart RS, Klein CJ, Wilson K, Steinback C, Lourival R, Kircher L, Possingham HP (2009) Marxan with zones: software for optimal conservation based land- and sea-use zoning. Environ Model Softw 24:1513–1521

Wilson KA, Underwood EC, Morrison SA Klausmeyer KR, Murdoch WW, Reyers B, Wardell-Johnson G, Marquet PA, Rundel PW, McBride MF, Pressey RL, Bode M, Hoekstra JM, Andelman SJ, Looker M, Rondinini C, Kareiva P, Shaw MR, Possingham HP (2007) Conserving biodiversity efficiently: what to do, where and when. PLoS Biol 5:e223. doi:10.1371/journal.pbio.0050223

Wilson KA, Carwardine J, Possingham HP (2009) Setting conservation priorities. The year in ecology and conservation. Ann N Y Acad Sci 1162:237–264

Wilson KA, Meijaard E, Drummond S, Grantham HS, Boitani L, Catullo G, Christie L, Dennis R, Dutton I, Falcucci A, Maiorano L, Possingham HP, Rondinini C, Turner W, Venter O, Watts M (2010) Conserving biodiversity in production landscapes. Ecol Appl 20:1721–1732. doi:10.1890/09-1051.1